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Intertidal wetlands of Port Curtis

Ecological patterns and processes and their implications

Rod M. Connolly, David R. Currie, Karen F. Danaher, Malcolm Dunning, Alistair Melzer, John R. Platten, Damon Shearer, Peter J. Stratford, Peter R. Teasdale, Maria Vandergragt

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CRC for Coastal Zone Estuary & Waterway Management

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Intertidal wetlands of Port Curtis: ecological patterns and processes, and their implications

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Executive summary

Port Curtis (the Port) is an outstanding natural harbour in and adjacent to the Great Barrier Reef Marine Park. The Port is heavily industrialised around the city of Gladstone along the western shoreline, but elsewhere retains large tracts of natural intertidal wetlands. The overall objective of the Intertidal Wetlands study was to describe ecological patterns and processes and determine the importance of different intertidal wetland habitats in Port Curtis. Considerable effort also went into developing an illustrative conceptual understanding of processes and threats in the area.

A mapping survey of mangroves, saltmarsh, saltflats, mudflats, seagrass and intertidal rocky substrate in Port Curtis combined data from Landsat satellite imagery, aerial photography and field inspections. The intertidal wetlands of Port Curtis are characterised by strong zonation. Saltflats without vegetation (landward of mangroves) cover 18% of the intertidal area. At the edge of these saltflats, where conditions are less saline, samphires and saltmarsh grasses occur, though they occupy only 4% of the intertidal area. Mangroves cover 31% of the area. Unvegetated mudflats (seaward of mangroves) cover 25% of the area. Seagrass meadows were estimated to cover 20% of the area, although this probably overestimates seagrass cover for much of the year. Cover within defined seagrass areas is patchy and highly seasonal, and the estimate was made at the period of peak seagrass cover.

Much of the seagrass in Port Curtis is close to existing or proposed port infrastructure and dredged channels. Although the seagrass is apparently healthy, the location of meadows leaves them vulnerable to direct impacts of future port infrastructure developments such as wharves, breakwaters and reclamation. The seagrass is of regional importance as one of the few large areas of seagrass in central Queensland, with the closest other areas more than 170 km to the north and south.

The Shoalwater Coast bioregion (which includes the Curtis Coast) has the second largest proportion of saltmarsh/saltflats of any estuarine area in Queensland, presumably because of its aridity (average annual rainfall only slightly over 1000 mm). Over 1600 hectares of tidal wetlands, particularly saltflats, were lost from Port Curtis between 1941 and 1999, due to large-scale reclamation for industrialisation, urbanisation and port development. Dietary analysis of juvenile fish shows that some species of fish (e.g. mullet) feed on saltflats, so the loss of this habitat could have an adverse effect on fisheries sustainability.

The chemical characteristics of mudflat, mangrove, saltflat and seagrass sediments were compared to determine associations between sediment parameters and habitat types. Physicochemical parameters were strongly correlated with, and probably determine, zonation of intertidal vegetation. Salinity of sea water in sediment (pore water) was the most important parameter, maintaining the extensive hypersaline saltflats landward of the main mangrove stands. At the land–sea margin, freshwater runoff dilutes porewater salinity sufficiently to allow a narrow strip of mangroves to take over from saltflat habitat, but not so much as to allow colonisation by less salt-tolerant terrestrial trees. Any reduction in freshwater flows resulting from increased development, land infill or climate change would therefore alter the vegetation composition and distribution in Port Curtis.

Recreational fishing is very popular and has a very high participation rate in Gladstone. The state of recreational fisheries is one indicator of environmental health and ecological productivity. Catch rates of whiting are lower in Port Curtis than at Cape Capricorn situated immediately outside the Port. Both locations show a decline in catch rates over time, but over the last ten years this trend has been more obvious inside the Port than at Cape Capricorn. No significant trend was established for offshore catches.

The current study provides evidence that freshwater flows are important to the productivity of Port Curtis. Years of large flow tend to have higher benthic invertebrate productivity, resulting in higher growth rates in fish such as whiting, which are caught earlier in their life cycle than they would otherwise be. This relationship between freshwater flows and fisheries productivity has been shown in several other places in the world, although the relationship does not hold true in all situations. The influence of flow on catch rates probably does not surprise local stakeholders but the immediacy of the effect (i.e. absence of a lag effect) is surprising. Reductions in freshwater flows as a result of increased water extraction for industry or urban use, or as a result of climate change, are likely to have an adverse effect on fisheries sustainability.

A trawl survey of 105 intertidal and shallow subtidal sites showed that fish assemblages of Port Curtis are diverse, with 88 species of fish recorded. The catch was dominated numerically by two small schooling species, ponyfish (*Leiognathus equulus*) and herring (*Herklotsichthys castelnaui*), common in inshore coastal waters elsewhere in tropical and subtropical waters of Australia. Fish assemblages change gradually from the more estuarine waters of The Narrows to open coastal waters. In addition, assemblages were distinguishable in different habitats, particularly seagrass meadows, and in varying water depths.

Seagrass meadows are less conspicuous in Port Curtis than in many other large estuarine embayments along the Australian east coast. They are nevertheless of great importance ecologically, particularly for fish. First, seagrass was shown to support a unique fish assemblage, thus promoting local biodiversity. Second, studies using stable isotope analysis to trace energy pathways and nutrient cycling demonstrated that seagrass is also important at the base of food webs beyond the seagrass meadows themselves. The food webs that sustain the many economically important fisheries species caught predominantly over mudflats (e.g. whiting) or in mangrove-lined creeks (e.g. mud crabs) rely largely on organic matter produced in seagrass meadows. They mostly do not consume the seagrass organic matter directly, but prey on small crustaceans and worms that do, either directly or via detritus. Despite seagrasses covering a smaller area of the Port than do mudflats or mangroves, they contribute more to the food webs that sustain fisheries species living over mudflats.

This is contrary to earlier ecological theory that mangroves drive subtropical estuarine food webs. While mangroves may be important to the animals living in the mangroves themselves, they do not play a major role in the provision of food for fisheries species occurring in other habitats in Port Curtis. It is important to note, however, that mangroves do provide other ecosystem benefits by stabilising and oxygenating sediments, and providing habitat structure for juveniles of economically important species such as banana prawns. Over the last decade, a paradigm has emerged that in areas dominated by large mudflats, microalgae living on and in the mud (benthic microalgae) are important in food webs. This has been suggested for the Fitzroy estuary just to the north of The Narrows. In the subtropical waters of Port Curtis, however, microalgae appear to play only a minor role in the provision of food for the fisheries species studied here.

A conceptual understanding of ecological patterns and processes maintaining coastal wetlands in Port Curtis was developed in interviews with scientists and other stakeholders. This led to a series of conceptual diagrams that summarise current knowledge about intertidal wetlands in this subtropical port. Five attributes or processes were identified as being of great importance: 1) tidal water movement, 2) freshwater runoff, 3) trophic (feeding) links, 4) severe storms and 5) droughts. The five top pressures impacting the wetland ecosystem were identified as: 1) land infill, 2) dredging, 3) barriers and dams, 4) contaminants (including plant nutrients) and 5) climate change.

This report can assist in the identification of management measures, such as protocols to limit dispersal of sediments during dredging for port development. The habitat maps provide a detailed summary of the distribution of the intertidal wetlands in Port Curtis, and can play an immediate role in planning and decision making in the region. The trade-off between port development and management for biodiversity, at sites where development is designated to occur, requires an understanding of the importance of the wetland ecosystems. This report includes scientific criteria required for determining the value of the intertidal wetlands of Port Curtis, and can direct the focus for future research aimed at better informing decisions about the relative significance of the wetland habitats.

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Chapter 1. Introduction

Rod M. Connolly, Karen F. Danaher and Alistair Melzer

1.1 Background

Port Curtis is an outstanding natural harbour supporting extensive industrial development along the western shoreline and having large tracts of natural intertidal wetlands elsewhere. Plans for further coastal development and port extension are well advanced, yet the port is adjacent to, and in some places inside, the Great Barrier Reef (GBR) World Heritage Area. The stakeholder landscape is dominated by industrial groups and there are good levels of awareness of the need for protection and/or replacement of wetlands. Academic and government research institutions have become active only recently in Port Curtis, and probably because of this there have been fewer scientific studies of wetlands here than in bays close to other major coastal cities in Australia. As a result, the values of intertidal wetlands in Port Curtis are largely unknown (Melzer & Tanner 2005). Although recreational and commercial fishing are of obvious importance to the local community, basic ecological and economic attributes of fisheries species associated with wetlands have not been quantified.

The overall objective of the Intertidal Wetlands study was to determine the ecological functions and importance of the different intertidal wetland habitats of Port Curtis. Investigations focused on the soft-sediment intertidal wetland habitats in Port Curtis, namely: saltflats, saltmarsh, mangroves, mudflats and seagrass. The roles of these intertidal habitats in ecological processes in deeper parts of the port were also addressed. The descriptive ecology in this report provides useful information about intertidal wetland processes, structures and distributions to contribute to the development of management options for the maintenance and protection of wetlands.

1.2 Need

The ecological, social and economic values of Port Curtis are largely unknown. While industrial operations in the area have increased, so too has awareness of the need to protect and/or replace the wetlands that characterise the port area. The more recent involvement of research institutions and government agencies has propelled the search for baseline values and attributes of the Port Curtis environment, in a bid to address impacts of development on the diverse and extensive intertidal wetlands. Many of the issues addressed in this report arose from the State of Port Curtis conference in October 2002, attended by a diverse array of stakeholders.

The State of the Environment Report for 2003 (EPA 2003) states that despite efforts since 1999 to improve coastal management responses, coastal zone condition is not improving significantly and continues to decline against a number of criteria. Pressures on the coast highlighted in the Curtis Coast Regional Coastal Management Plan include wetland clearing, reduction in freshwater flows at the mouth of the Boyne River due to the Awoonga Dam, and changes to landscape character due to vegetation disturbance for residential and industrial development.

The Curtis Coast Regional Coastal Management Plan also lists the following threats to coastal wetlands in the Curtis Coast Region (Environmental Protection Agency 2003):

- reclamation associated with port, industrial and urban development
- infrastructure development changing surface water flows to coastal wetlands
- urban, industrial and agricultural sites discharging pollutants or water of low quality
- shipping and related activity (potential spills)
- mining
- dredging operations and sediment inputs from land-based sources increasing turbidity, which can impact on seagrass beds through reduced light
- saltflats being used inappropriately as sites for re-establishment of mangrove habitat
- catchment degradation contributing to increased sedimentation and runoff.

The impacts of these significant inputs and pressures from the upper catchment and from within the estuarine area itself are poorly understood in Port Curtis, and there are critical knowledge gaps regarding their effects on the ecological health of the estuary and adjacent marine areas. The current intertidal wetlands research synthesises information on the ecological characteristics of Port Curtis, providing baseline mapping of vegetation and ecological data required to assist local managers and regulators with improved assessment of coastal development and management pressures. The project also provides scientists and other stakeholders with an improved understanding of the ecological roles and functions of their local estuarine environment. Some of the findings in this report also have generic wider relevance to other subtropical intertidal wetlands. The outcomes of this project are intended to provide a foundation for ecological valuations of intertidal wetlands in Port Curtis, and for the development of management options for the area.

1.3 Objectives

The objective of this study was to describe ecological patterns and processes and determine the importance of different intertidal wetland habitats in Port Curtis. Further, considerable effort went into developing the conceptual understanding of processes and threats in Port Curtis, and these led to a series of conceptual diagrams which summarise knowledge of intertidal wetlands in Port Curtis.

The research provides rigorous baseline data on ecological patterns and processes that will underpin future management strategies in Port Curtis.

1.4 Report overview

This report takes the form of a series of chapters by individual research teams, each chapter providing an introduction, methods, results and discussion.

- **Chapter 1** provides the background to the Port Curtis Intertidal Wetlands project, and describes the Port Curtis intertidal environment on which the subsequent chapters are based.

- Coastal mapping showing the locations and extent of the mangroves, saltmarshes, seagrass, exposed banks and rocky substrate is presented in **Chapter 2**, which also discusses the regional significance and current condition of the first three of these habitats.

- **Chapter 3** compares sediment physicochemical measurements among four habitat types (mudbanks, mangroves, saltflats and seagrass) within Port Curtis, and determines the principle sediment parameters associated with a given habitat type.

- **Chapter 4** describes recreational fishing trends in Port Curtis, by comparing catch rates in locations within the Port to those from offshore waters. This chapter also discusses the relationship between the catch rate of fish and macrobenthos productivity, and the effect of freshwater flows on catch rates.

- As estuaries are widely recognised as important nursery grounds for juvenile fish, **Chapter 5** describes demersal fish species composition and distribution in Port Curtis. The importance of structural complexity in influencing species composition, and the importance of environmental variables in governing the abundance and diversity of fish are also examined.

- **Chapter 6** reports trophic pathways sustaining fisheries species, especially those occurring over shallow mudflats in Port Curtis. This chapter also highlights the importance of intertidal habitats as a trophic source for animals that do not necessarily reside *in situ*.

- **Chapter 7** provides conceptual diagrams of the key ecological processes and anthropogenic pressures in Port Curtis, which also has wider relevance to other subtropical port environments. This chapter summarises and illustrates the information and knowledge currently available for the Port Curtis region so that it can be understood by scientists, industrial stakeholders and the general community.

- **The final chapter** synthesises the broader findings of the report and their implications for Port Curtis, and identifies critical knowledge gaps that require further research.

1.5 The study area

Port Curtis is located on the central Queensland coast (Figure 1.1). The Port is the result of former river valleys being submerged by rising sea levels during the Quaternary period (Conaghan 1966). Sediments supplied by rivers and offshore sources are slowly infilling the valleys (QDEH 1994). At present, the outer series of barrier islands (Curtis and Facing) provides protection for enclosed waters, allowing estuarine environments to establish. Most of these estuaries receive a limited supply of fresh water from a narrow coastal hinterland.

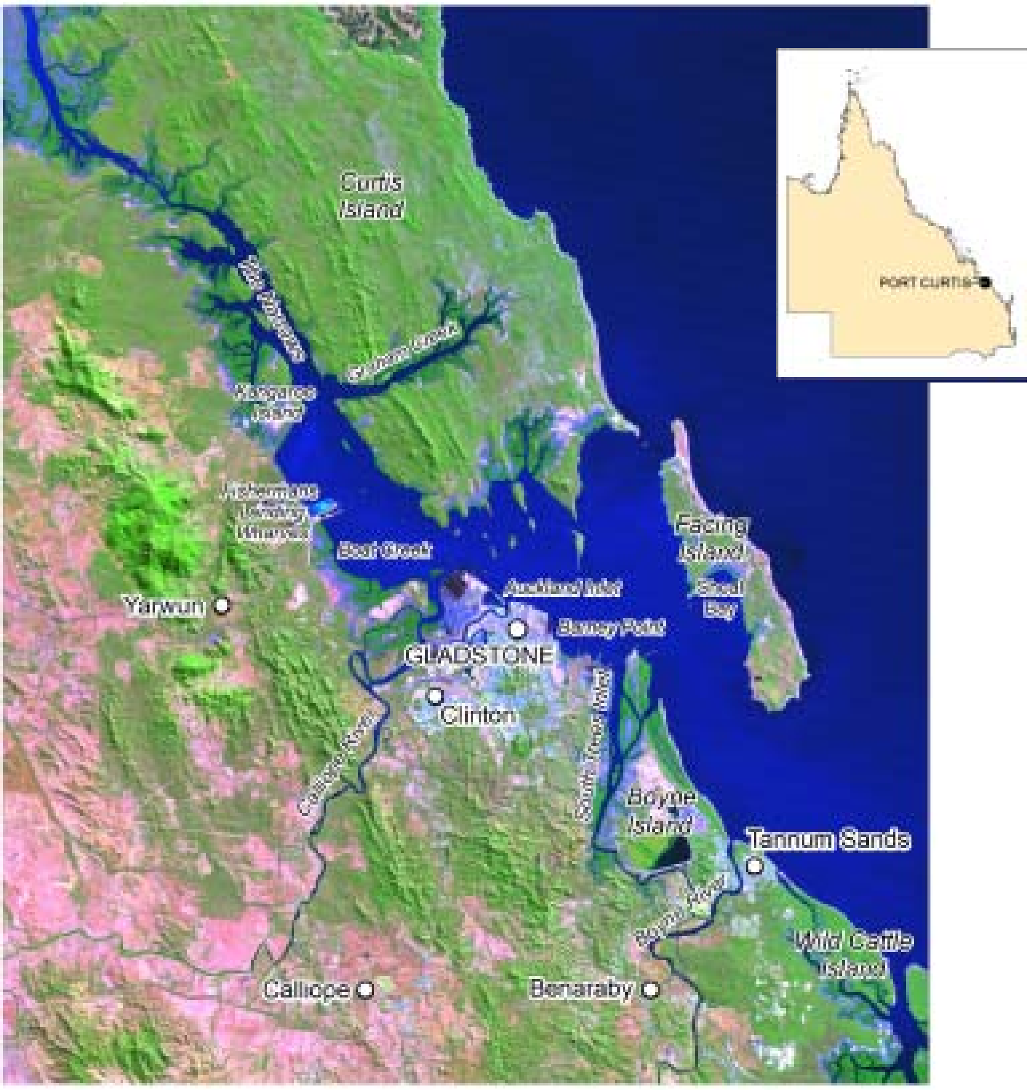

Figure 1.1. Site map of the study area at Port Curtis

Port Curtis experiences a subtropical, maritime climate. The average minimum and maximum ambient temperatures for the study area range from about 23–29°C in summer to 11–20°C in winter (BOM 2005). Water temperatures vary from 18°C in winter to 29°C in summer. The majority of Gladstone's annual average 1020 mm rainfall is received during the summer months. Evaporation rates are high and generally exceed rainfall, with the average annual evaporation rate for Gladstone being 1748 mm. The predominant winds are the southeasterly trade winds. Tropical cyclones occur approximately every 7 years (BOM 2005). Tides are semi-diurnal with a maximum amplitude of 4.69 m at Gladstone, rising to 6 m at Ramsay's Crossing in the Narrows (Maritime Safety Queensland 2004). Many of the estuaries have multiple entrances, so tidal circulation patterns are complex (QDEH 1994). The region falls within the Shoalwater Coast bioregion as defined in the *Interim marine and coastal regionalisation for Australia* (IMCRA Technical Group 1998).

Port Curtis is a major industrial centre, with industries including an aluminium refinery and smelter, a cement production works, chemical plants and a power station. The harbour is home to Queensland's largest multi-commodity port. The intertidal wetlands have been impacted upon by industrial development, port development, harbour dredging, discharge of effluent and extensive reclamation. Around Gladstone City wetland habitats have been extensively cleared, filled or modified (Duke *et al*. 2004). Further port and industrial development is planned and is likely to be accompanied by increasing urbanisation.

1.6 Intertidal habitats of Port Curtis

The Curtis Coast region comprises a complex range of coastal habitats that includes extensive areas of intertidal wetlands (Table 1.1).

Table 1.1. Typical habitat zones in intertidal areas of the Port Curtis region

Intertidal wetland	Notes
Algal mats	Occur as green filamentous algae on mudbanks or cyanobacterial mats on saltflats
Mangrove	No universally accepted definition, but considered intertidal trees and shrubs (Bruinsma 2000)
Mudflat	Unconsolidated intertidal sediments seaward of the mangroves lacking conspicuous vegetation
Rocky shores, fringing coral reefs, rocky reefs, rubble and sand beaches	Along the edges of continental islands and sheltered areas, support diverse marine flora and fauna, but are not the focus of this report
Saline grassland and Casuarina woodlands	Habitat at the border of intertidal and true terrestrial environment, dominated by salt couch and *Casuarina glauca*
Saltflat	Hypersaline, unvegetated flats shoreward of saltmarsh habitat
Saltmarsh	Intertidal plant communities dominated by salt-tolerant herbs and low shrubs and saltmarsh grass (Bruinsma 2000)
Seagrasses	Intertidal and subtidal meadows on mudflats/mudbanks

Mangroves are widespread along estuarine reaches of most creeks and rivers. In many places, saltmarshes occur landward of the mangroves and these in turn are backed by saltflats. Intertidal mudflats and intertidal and subtidal seagrass beds are found in creek curves and river banks where tidal flows influence sediment deposition, as well as across much of the Port area. Rocky and sandy intertidal habitats are found along inshore continental islands.

Mangrove, saltmarsh and seagrass communities are recognised for their value to fisheries production, and directly support local inshore and offshore fisheries through the provision of food, shelter, breeding and nursery areas. Quinn (1992) has estimated that estuarine habitats in Queensland are critical to more than 75% of commercially and recreationally important fish and crustacean species during some stage of their life cycle.

1.6.1 Mangroves

Mangroves are a diverse group of predominantly tropical shrubs and trees growing in the marine tidal zone (Duke 1992). These marine plants are thought to serve a wide variety of functions (Claridge & Burnett 1993; Ewel *et al*. 1998) including:

- physical protection of the coastal fringe from erosion and flooding
- sediment trapping
- primary production, nutrient uptake and transformation
- provision of food, shelter, breeding and nursery areas for a wide variety of marine and terrestrial animal species.

At a regional scale, the distribution of mangrove species is determined by a number of factors including temperature, rainfall, catchment area and tidal inundation. Mangrove species are limited in their latitudinal distribution by their physiological tolerance to low temperatures (Duke *et al*. 1998). The majority of mangrove species are limited to tropical environments where the mean winter temperatures are above 20^{o}C. Consequently, mangrove species diversity generally decreases with increasing latitude. In Queensland this phenomenon can be clearly seen along the east coast, with Cape York recording 36 species, the Curtis Coast region recording 14, and southeast Queensland recording nine species (Duke 1992).

Mangrove species are also variable in their tolerance to the variety of environmental parameters experienced in the intertidal zone, including salinity, soil type, frequency of inundation (both tidal and fresh) and wave action. Accordingly, mangrove species distribution within an estuary can generally be related to the variation of these factors and typical mangrove zones often result. For example, closed *Rhizophora* zones (or communities) within Queensland generally occur on the water's edge where they receive inundation with every high tide. In contrast, open or closed *Ceriops* communities, which occur towards the landward mangrove edge, are generally only inundated on the spring tides that occur once or twice per month. Fourteen species of mangroves are reported to occur in Port Curtis (Saenger 1996). These are listed in Table 1.2.

Table 1.2. Mangrove species recorded in the Curtis Coast region (Saenger 1996)

Mangrove species	Common name
Acanthus ilicifolius L.	Holly leaf mangrove
Acrostichum speciosum Willd.	Mangrove fern
Aegialitis annulata R. Br.	Club mangrove
Aegiceras corniculatum (L.) Blanco	River mangrove
Avicennia marina (Forsk) Vierh.	Grey mangrove
Bruguiera exaristata	Orange mangrove
Bruguiera gymnorrhiza L. Lam.	Large-leafed orange mangrove
Ceriops tagal C. T. White	Yellow mangrove
Excoecaria agallocha L.	Milky mangrove
Lumnitzera racemosa Willd.	Black mangrove
Osbornia octodonta F. Muell.	Myrtle mangrove
Rhizophora stylosa Griff.	Red mangrove
Xylocarpus granatum Koen	Cannonball mangrove
Xylocarpus moluccensis Pierre	Cedar mangrove

1.6.2 Seagrasses

Seagrasses are productive flowering plants able to complete their life cycle completely or partly submerged in marine waters (Mateer 1998). Coastal and surface topography, water depth and turbidity, and freshwater runoff all influence seagrass distribution and abundance patterns.

The fisheries value of seagrass habitat as nursery grounds for juvenile commercial fish and prawn species in Queensland is well documented (e.g. Rasheed *et al*. 1996; Watson *et al*. 1993). Seagrass also provides food for dugong and green sea turtles, with both these species observed within Port Curtis during the 2002 baseline seagrass survey (Rasheed *et al*. 2003). Seagrass beds are vulnerable to sedimentation, salinity levels and water pollution (Coles & McKenzie 2004).

In Australia, seagrasses grow in temperate and tropical waters. In Queensland, the highest number of species is found near the tip of Cape York, with a gradual decline in the number of species southward along the east coast (Coles *et al*. 1989; Poiner *et al*. 1989). Significant seagrass beds are found at many locations throughout the Curtis Coast region, with intertidal and subtidal communities exhibiting generally different species compositions and canopy coverage (Currie *et al*. 2003). There are six species of seagrasses found in the Curtis Coast region (Rasheed *et al*. 2003). These are listed in Table 1.3.

Table 1.3. Seagrass species recorded in the Curtis Coast Region (Rasheed *et al.* 2003)

Seagrass species
Halodule uninervis (Forsk.) Aschers
Halophila decipiens Ostenfeld
Halophila minor (Zoll.) den Hartog
Halophila ovalis (R. Br) Hook. F.
Halophila spinulosa (R. Br.) Aschers.
Zostera capricorni Aschers.

1.6.3 Saltmarshes

Saltmarshes are an intertidal habitat dominated by salt-tolerant herbs and low shrubs, such as samphires and saltmarsh grass. In Port Curtis, saltmarsh occurs at the seaward edge of extensive saltflats, usually just landward of mangroves. It can also occur at the terrestrial side of saltflats where freshwater input reduces salinity. The vegetation consists of sparse ground cover of species such as *Suaeda* spp., *Sarcocornia quinqueflora* and *Sporobolus virginicus*. In contrast to mangrove species, saltmarsh species diversity and community complexity in Queensland increases with increasing latitude (Zeller 1998). Saenger (1996) records 40 saltmarsh species (shrubs, grasses, herbs and algae) for the Curtis Coast region.

Although saltmarsh environments are generally only inundated with the high tides they can play an important role as fisheries habitat. Specifically, it has been shown in southeast Queensland that shallow tidal pools within the saltmarshes provide transitory feeding habitat for larval and juvenile fishes (Zeller 1998). Connolly (1999) studied the use by fish species of subtropical saltmarsh habitat in southeast Queensland and confirmed that both vegetated and unvegetated saltmarsh habitats are utilised by a surprisingly high number and diversity of both estuarine-resident and estuarine-marine fish species. More than half of the fish species caught on the saltmarsh habitat were of direct economic importance, and several of these species were common without dominating the catch numerically. The distribution of fish on saltmarshes was found to be most strongly influenced by proximity to intertidal, mangrove-lined feeder creeks, with more species and more individuals close to creeks than further away (Connolly 2005).

Table 1.4. Saltmarsh species recorded in the Curtis Coast region (Saenger 1996)

Shrubs	
Diospyros ferrea var. *geminata*	*Salsola kali*
Enchylaena tomentosa	*Sarcocornia quinqueflora*
Halosarcia halocnemoides	*Suaeda arbusculoides*
Halosarcia inica var. *leiostachyum*	*Suaeda australis*
Myoporum acuminatum	
Grasses and herbs	
Aristida calycina	*Limonium australe*
Branchyscombe basaltica	*Paspalidium gracile*
Bulbostylis barbarta	*Portulaca napiformis*
Cyperus polyustachyos	*Scirpus litoralis*
Cyperus scariousus	*Sesuvium portulacastrum*
Eleocharis geniculata	*Spergularia media*
Emilia sonchifolicacus	*Spergularia rubra*
Epaltes australis	*Sporobolus virginicus*
Eragrostis elongata	*Veronia cinerea*
Fimbristylis ferruginea	*Vittadinia triloba*
Fimbristylis punctata	
Algae	
Anabaena torulosa	*Lyngbya aesturii*
Anacystis marina	*Microcoleus lyngbyaceus*
Calothrix crustacea	*Oscillatoria nigro-viridis*
Chroococcus turgidus	*Phormidium angustissimum*
Hormidium subtile	*Thizoclonium capillare*

1.6.4 Saltflats

Saltflats are hypersaline, unvegetated areas high in the intertidal zone, inundated only at high spring tides (Saenger 1996). They are characterised by poorly drained clay soils, high evaporation rates and a low, strongly seasonal rainfall (Saenger 1996). The surface of the saltflats is devoid of vascular plants, but can be covered by a thick algal mat. The mat combines with the top layer of clay to form a leathery surface which peels off and cracks into sheets as the saltflat dries (Olsen *et al*. 1980).

Increasing pressure by recreational use of off-road vehicles has been identified in some saltflat areas (Melzer & Tanner 1995). Regional studies often report a paucity of flora and fauna in saltflat habitat (WBM 1990), but offer little detail about their ecology, importance or status.

1.6.5 Mudflats

Intertidal mudflats lacking conspicuous vegetation form the most extensive habitat in Port Curtis. They occur where tidal flows deposit sediments (silt and sand). Macrobenthos in and on mudflats are sometimes regarded as indicators of ecosystem health because of their close association with sediment. Recent studies have investigated the relationships among macrobenthos and physical and chemical features of the Curtis Coast region, especially in mudflats. Studies including investigations of intertidal mudflats have illustrated strong links between macrobenthos and environmental and sediment characteristics (Currie & Small 2005). Mudflats lacking conspicuous vegetation may play an important trophic role yet to be fully acknowledged. Erftemeijer and Lewis (1999) recognised that intertidal mudflats constitute an important habitat that support a high biodiversity and biomass of benthic invertebrates, sustain productive fisheries and provide important feeding grounds for migratory shorebirds.

1.6.6 Rocky and coralline intertidal habitat

Rocky foreshores provide a hard substrate for the attachment of algal flora as well as the long-term attachment of sedentary invertebrates such as barnacles, oysters and tube worms (Zeller 1998). Both macro- and microalgae can play a key role in primary production (Alongi 1998).

A few locations within the Curtis Coast region support rocky intertidal areas, although few regional studies mention these habitats or associated fringing coral reefs. Some research on intertidal rocky shore invertebrate communities has been reported from Keppel Bay and Bundaberg coastal regions, in habitats similar to those in the Curtis Coast region, and data exists also for Curtis Island (Kay & Coates 2002). These hard-surface, intertidal shores are not the focus of this report but are included in the mapping of habitats in Chapter 2.

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Chapter 2. The intertidal wetlands of Port Curtis

Karen F. Danaher

2.1 Introduction

2.1.1 Project scope

The objective of this component of the intertidal wetlands study was to produce detailed maps of all of the intertidal wetlands of Port Curtis, specifically the mangroves, saltmarsh, seagrass, intertidal mudbanks, sandbanks and rocky substrates. This has been published with detailed maps as Danaher *et al*. (2005). The procedure, summary results and relevance of the mapping are reported here.

2.1.2 Coastal wetland mapping programs of Queensland's DPI&F

In order to manage marine vegetation communities which are protected under the *Queensland Fisheries Act 1994*, the Queensland Department of Primary Industries and Fisheries (DPI&F) has undertaken two mapping programs. The Queensland Coastal Wetlands Resource Mapping Project has mapped all of Queensland's mangrove and saltmarsh communities at a scale of 1:100 000. Appendix 2.1 lists the reports of this mapping program. The techniques devised in that project were expanded in this Coastal CRC project to map the mangrove and saltmarsh communities of Port Curtis in more detail at a scale of 1:25 000. This scale allows for more accurate monitoring and change detection at the local level in the future.

DPI&F, in conjunction with the CRC for the Great Barrier Reef World Heritage Area (Reef CRC) and Queensland's port authorities, are mapping and monitoring seagrass communities along the Queensland coast. For Port Curtis, baseline mapping was completed in 2002. This mapping has been incorporated into the current Port Curtis intertidal wetlands mapping work. For full details on this baseline dataset refer to Rasheed *et al.* (2003).

2.2 Methods

2.2.1 Mangroves and saltmarshes

Data

The map of mangrove and saltmarsh communities was produced from Landsat imagery, aerial photography and field data. A Landsat 7 Enhanced Thematic Mapper (ETM+) satellite image was captured on 24 July 2002. This imagery was obtained from the Queensland Department of Natural Resources, Mines and Water's Statewide Land Use and Trees Study (<www.nrm.qld.gov.au/slats>) and was already radiometrically corrected and geometrically rectified to the Australian Map Grid (Zone 56) using the Australian National Spheroid and the Geodetic Datum of Australia 1994 (Armiston *et al*. 2002). In addition to the satellite imagery, aerial photography was used, and is listed in Table 2.1.

Table 2.1. Aerial photography datasets used

Organisation	Program	Scale	Date
Beach Protection Authority	St Lawrence to Hervey Bay	1:12 000	2001
Dept Natural Resources, Mines & Water	Gladstone	1:25 000	1999, 1996
Gladstone Port Authority		1:40 000	2000
Gladstone Port Authority		1:5 000	2003

Mapping methods

The satellite imagery was processed using ERDAS Imagine ® 8.3.1 on a personal computer with a Microsoft Windows XP operating system. The ETM+ panchromatic band, which is captured with 15 m x 15 m pixels, was merged with six of the ETM+ multispectral bands (excluding Band 6, the thermal band) to improve the spatial resolution from 30 m x 30 m pixels to 15 m x 15 m pixels. The six new bands were contrast stretched using a linear stretch and breakpoints to highlight the intertidal regions. In order to limit the area of the classification to the coastal wetland environments, the terrestrial land features were masked out manually. The upper limit of the intertidal zone was identified using a false colour composite of ETM+ bands 1, 4 and 5 (through blue, green and red colour guns, respectively) in conjunction with colour aerial photography, topographic maps and fieldwork. Deep water bodies seaward of the mangroves and deep, permanent (non-tidal) pools were spectrally masked out using an ETM+ band 4 (near infrared) image.

The remaining imagery, which included the intertidal zone and a small strip of adjacent coastal land, was processed using an unsupervised classification procedure. ERDAS Imagine uses the Iterative Self-Organising Data Analysis

Technique (ISODATA) classification algorithm in order to create clusters of pixels that are spectrally similar. The ISODATA utility repeats the clustering of the image until either a maximum number of iterations has been performed, or a maximum percentage of unchanged pixels (convergence threshold) has been reached between two iterations (ERDAS 1997). A limit of 30 iterations or a convergence threshold of 99% was set in this classification. The resulting classes were labelled according to their dominant cover type with the aid of the higher resolution aerial photography. Manual interpretation of the aerial photography was also used to delineate mangrove and saltmarsh classes that were not identified during the digital classification. To help map interpretation, clumps of pixels less than 1000 m^2 were eliminated and the classification image was smoothed using a 3 x 3 pixel moving kernel.

Mangroves were classified to the community level on the basis of dominant genus present and relative densities of the whole community. The density of the community was determined by estimating the projective foliage cover (PFC). A canopy cover of greater than 50% was classified as closed, while less than 50% was identified as open. The standard Specht (1987) vegetation categories of ‘forest’ and ‘shrub’, which are based on height, were not included in this classification. This is due to the fact that community height cannot be determined from the Landsat ETM+ data. Only areas subject to tidal inundation were included in this mapping exercise. Excluded classes that occurred within the intertidal zone included permanent (non-tidal) pools of water and elevated land containing terrestrial vegetation.

Field methods

Field data were collected during February and June, 2004. Field data collected for the Queensland Coastal Wetlands Mapping Project (De Vries *et al*. 2002) and the Calliope River Study (McKinnon *et al*. 1995) were also used. Sites were selected to represent most classes and were replicated over the study area. These were accessed either by boat, four-wheel-drive vehicle or on foot. Information on mangrove community floristics and structure was documented at selected sites. Data recorded included the specific composition of mangroves, dominant genus, estimated density (PFC) of each vegetation layer, composition and hardness of substrate, and presence/absence of seedlings, samphires, grasses, algae, leaf litter, roots, ferns, epiphytes, sedges and ponds. A global positioning system (GPS) was used to determine the latitude and longitude of each field site, with an accuracy of ≤ 20 metres. Figure 2.1 shows the location of the field sites.

2.2.2 Exposed banks (mud and sand) and rocky substrate

Data

The exposed banks and rocky substrate classes were produced from aerial photography and the zero metre contour from the Gladstone Boating Safety Chart (DOT 2002).

Mapping methods

The zero metre contour line from the Gladstone Boating Safety Chart was obtained as digital data from the Department of Transport, Queensland. The exposed banks and rocky substrate classes were manually interpreted from aerial photography flown at low tide using this contour as a base. No attempt was made to separate mud from sand substrates. Unfortunately the Landsat ETM+ image was not captured at low tide so these habitats could not be automatically classified. The exposed banks and rocky substrate classes were added to the mangrove and saltmarsh classification in ERDAS.

2.2.3 Seagrass

The methods used for the seagrass mapping of Port Curtis are described in more detail in Rasheed *et al*. (2003). It should be noted that these coastal seagrass classes are not completely intertidal. The lower boundary of the coastal seagrass mapping in Port Curtis was usually less than 5 m below mean sea level. Because the area of the subtidal coastal seagrass components of the meadows is small, the entire seagrass meadows have been left intact and presented in this map.

Data

In addition to helicopter and dive surveys, the seagrass mapping used aerial photography. These were the same datasets from the Beach Protection Authority and the Gladstone Port Authority that are listed in Table 2.1.

Mapping methods

The boundaries of intertidal seagrass meadows were determined from observations made from a helicopter hovering at 10–100 m around spring low tide when meadows were exposed. GPS positions defined meadow boundaries with fixes taken approximately every 50 m or when there was a change in boundary direction. Meadows were classified based on species composition, above-ground biomass and landscape type (e.g. isolated seagrass patches, aggregated seagrass patches, continuous seagrass cover). At random sites, the helicopter hovered within a metre and experienced observers identified the seagrass species present and conducted visual estimates of above-ground biomass. In subtidal areas habitat characteristics were determined by diving along transects at certain sites.

At all sites (both helicopter and diver), the position was recorded using GPS, and assessments of seagrass species composition, seagrass above-ground biomass, percent cover of algae, depth below mean sea level and sediment type were conducted.

All the data were entered into a GIS with rectified colour aerial photography. A total of 32 community types were identified, based on species presence, density and dominance. For the purpose of this combined intertidal wetland map, the 32 classes from Rasheed *et al*. (2003) were amalgamated into nine classes based on the dominant species and landscape type.

2.2.4 Intertidal wetland GIS creation

The mangroves, saltmarsh, exposed bank and rocky substrate classification was converted from raster to vector format using ArcGIS version 9.0 software. To improve cartographic presentation of the data, the jagged vector boundaries were smoothed using a polynormal approximation with exponential kernel (ESRI 2004). At 1:25 000, 1000 m^2 was set as the minimum mapping unit so polygons with areas less than this were excluded. The coverage was then converted to an ESRI shapefile. The nine seagrass classes were joined to this shapefile.

The shapefile was overlaid on the Landsat ETM+ panchromatic image. Maps were produced using ArcGIS Version 9 to be viewed at a scale of up to 1: 25 000 (as in Figure 2.4).

2.2.5 Classification accuracy

Mangroves, saltmarsh, exposed banks and rocky substrate

A reliability code was assigned to each polygon within the classification based on the amount of ground truth data available (i.e. high resolution aerial photography and field data). The reliability codes are combined for both the positional accuracy and the vegetation type accuracy and are shown in Figure 2.1 and described in Table 2.2.

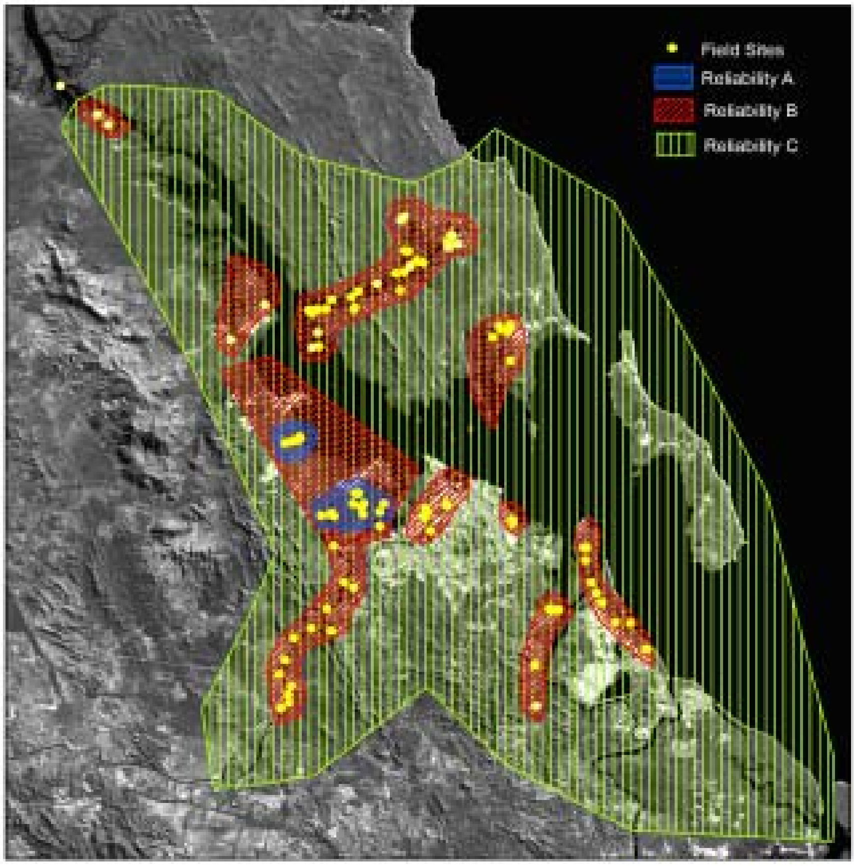

Figure 2.1. Field sites and reliability diagram for mangroves, saltmarsh, exposed banks and rocky substrate

Table 2.2. Reliability code and classification assigned to mangroves, saltmarsh, exposed banks and rocky substrate

Code	Reliability	Data
A	Highest reliability	Very high resolution aerial photography coverage, fieldwork conducted
B	High reliability	Very high resolution aerial photography coverage, no fieldwork **OR** High resolution aerial photography coverage, fieldwork conducted
C	Good reliability	High resolution aerial photography coverage, no fieldwork conducted

Seagrass

Seagrass meadow boundaries were determined generally from field surveys rather than interpreted from remote imagery. In some cases available aerial photography from the Gladstone Port Authority and Beach Protection Authority (see Table 2.1) was used to confirm the boundaries. The accuracy of the seagrass meadow boundaries depended on the range of mapping information and methods available for each meadow (Rasheed *et al*. 2003). The intertidal meadow boundaries followed with GPS had the highest accuracy and the submerged, subtidal areas had the lowest accuracy. Each meadow was assigned a mapping precision estimate (in metres). Mapping precision for coastal seagrass meadows ranged from ± 5 m for isolated intertidal seagrass meadows to ± 20 m for subtidal meadows. The mapping precision estimate was used to calculate a range of meadow area for each meadow and was expressed as a meadow reliability estimate in hectares. For the purpose of this combined intertidal wetland mapping, these reliability estimates were aligned to match the reliability codes of the mangroves, saltmarsh, exposed bank and rocky substrate classes. The reliability codes for the seagrass are described in Table 2.3.

Table 2.3. Reliability code and classification assigned to seagrass

Code	Reliability	Description
A	Highest reliability (mapping precision 5 m)	• Meadow boundaries mapped in detail by GPS from helicopter • Intertidal meadows completely exposed or visible at low tide • Relatively high density of mapping and survey sites • Recent aerial photography aided in mapping
B	High reliability (mapping precision 10 m)	• Meadow boundaries determined from helicopter and diver surveys • Inshore boundaries mapped from helicopter • Offshore boundaries interpreted from survey sites and aerial photography • Relatively high density of mapping and survey sites
C	Good reliability (mapping precision 20 m)	• Meadow boundary interpreted from diver surveys • All meadows subtidal • Relatively high density of survey sites • Recent aerial photography aided in mapping

The presence or absence of seagrass at each site was defined by the above-ground biomass. Where above-ground seagrass was absent, the presence of rhizome/root and seed-bank material was not reported. Survey sites with no seagrass can be found within meadows because seagrass cover within meadows is not always uniform and may be patchy and contain bare gaps or scars.

2.3 The distribution of the intertidal wetland communities

Table 2.4 gives the total area of the intertidal wetland community types at 1:25 000 in Port Curtis. Figure 2.2 shows the dominance by area of the intertidal wetland community types, and Figure 2.3 presents photographic examples of intertidal wetland habitats encountered. It should be noted that the occurrence and extent of seagrass meadows is more variable than mangrove and saltmarsh communities and may vary seasonally and between years. This seagrass survey was conducted at the time of year when the seagrasses were likely to be at their maximum distribution. While this distribution was likely to represent the maximum for 2002, the area may vary between years. DPI&F conducts annual monitoring for a subset of the meadows in Port Curtis. Figure 2.4 shows the aerial photography map of the contribution of each intertidal habitat type, for a section of the study area around Gladstone. Full maps are available in Danaher *et al*. 2005.

Table 2.4. Total area of intertidal wetlands for Port Curtis (selected examples shown pictorially in Figure 2.3)

Community	**Area (ha)**	**Approx. % of total**
Closed *Rhizophora*	4396	20.6
Closed *Avicennia*	100	0.5
Open *Avicennia*	85	0.4
Closed *Ceriops*	309	1.5
Open *Ceriops*	35	0.2
Closed *Avicennia/Ceriops*	745	3.5
Open *Avicennia/Ceriops*	13	0.1
Closed *Rhizophora/Avicennia*	350	1.7
Open *Rhizophora/Avicennia*	1	0.0
Closed *Aegiceras*	96	0.5
Closed *Aegiceras/Rhizophora*	38	0.2
Closed *Aegiceras/Avicennia*	23	0.1
Closed mixed mangroves	520	2.5
Dead *Rhizophora*	3	0.0
Dead *Ceriops*	14	0.1
Dead *Ceriops* with emergent *Avicennia*	8	0.0
Saltflat	3894	18.2
Samphire	486	2.3
Saline grass	193	0.9
Intertidal mud and sand banks (unvegetated)	5144	24.1
Exposed rocky substrates	297	1.4
Isolated *Zostera* patches	108	0.5
Aggregated *Zostera* patches	1807	8.5
Continuous *Zostera* cover	626	3.0
Isolated *Halodule* patches	25	0.1
Aggregated *Halodule* patches	1299	6.1
Continuous *Halodule* cover	245	1.2
Isolated *Halophila* patches		
Aggregated *Halophila* patches	391	1.8
Continuous *Halophila* cover		
Total	**21 250**	**100**

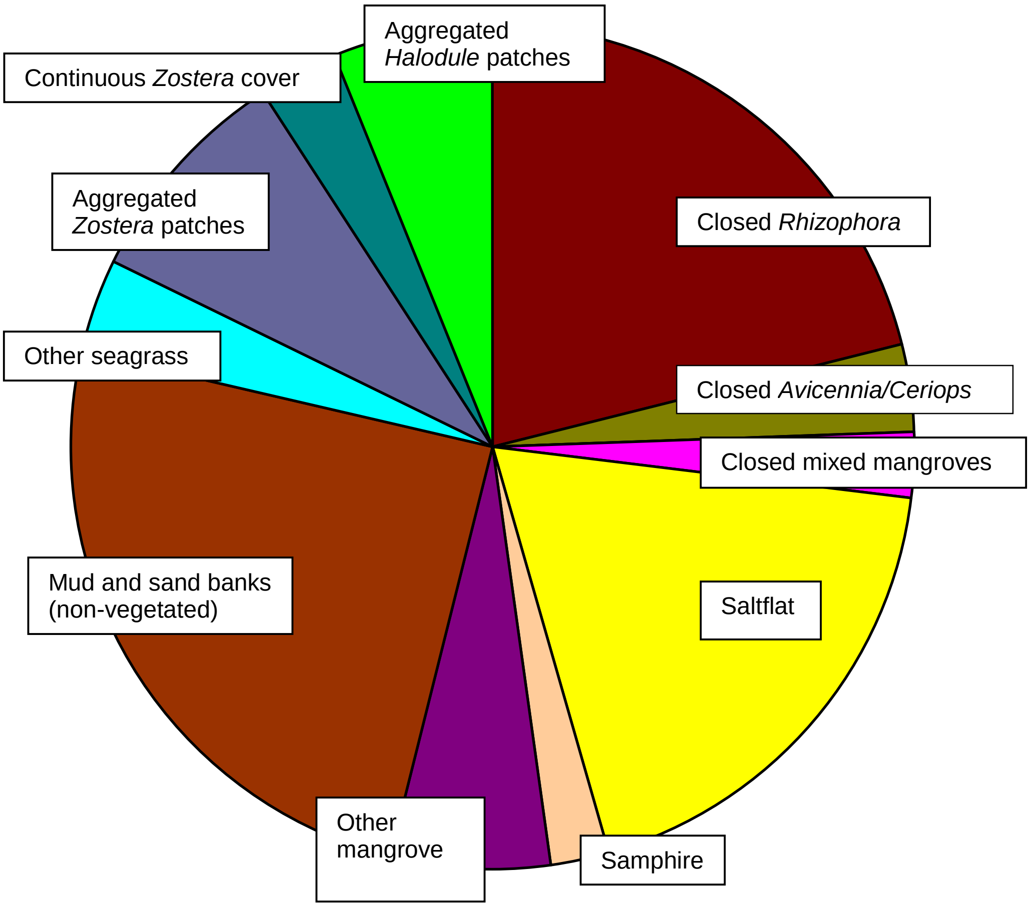

Figure 2.2. The dominance of the intertidal wetlands of Port Curtis by area

Continuous seagrass cover

An exposed mudbank at Barney Point

A saltflat at Barney Point

A closed *Rhizophora* community at Graham Creek

A closed *Avicennia/Ceriops* community at Boyne Island

Isolated seagrass patches

(Photographs courtesy of Karen Danaher)

Figure 2.3. Examples of intertidal wetland habitats

The intertidal wetlands of Port Curtis are characterised by strong zonation and extensive saltflats. Figure 2.5 shows their general distribution along the tidal profile. Mudbanks with or without seagrass occur seaward of all of the mangroves. The seaward edge of the mangroves is usually closed *Rhizophora* communities, around 6 m tall. *Avicennia* communities pioneering on the soft substrates seaward of these are not common. Behind the *Rhizophora* is usually a narrow band of *Avicennia/Ceriops* often only 10 m wide and less than 2 m tall. Then there are extensive saltflats with no vegetation. On some saltflats, where conditions are not as hypersaline, samphires and saline grass communities may develop. At the landward edge of the saltflats, a narrow band of *Ceriops* may occur. This community often contains *Lumnitzera racemosa* and *Excoecaria agallocha*. Along watercourses with freshwater input, closed *Rhizophora*/ *Avicennia* communities occur, while *Aegiceras* communities exist on accreting banks within winding watercourses. Upstream, with more freshwater input, mixed communities occur with many species, including *Xylocarpus moluccensis.*

Extensive areas of seagrass were found, mainly on the shallow coastal mud and sand banks, but rarely penetrating into subtidal areas off the edge of the banks (Rasheed *et al*. 2003). The intertidal seagrass communities on muddy sediments were dominated by *Zostera capricorni*. These are the most widely distributed communities within Port Curtis. The intertidal seagrass communities on sandy sediments were dominated by *Halodule uninervis*. These communities were found between Quoin and Facing Islands. They were also along the exposed coasts of Boyne and Wild Cattle Islands. *Halophila* dominated communities were located adjacent to Fisherman's Landing Wharves. *Halophila decipiens* and *Halophila spinulosa* dominated deeper areas (shallow subtidal and offshore more than 5 m below mean sea level).

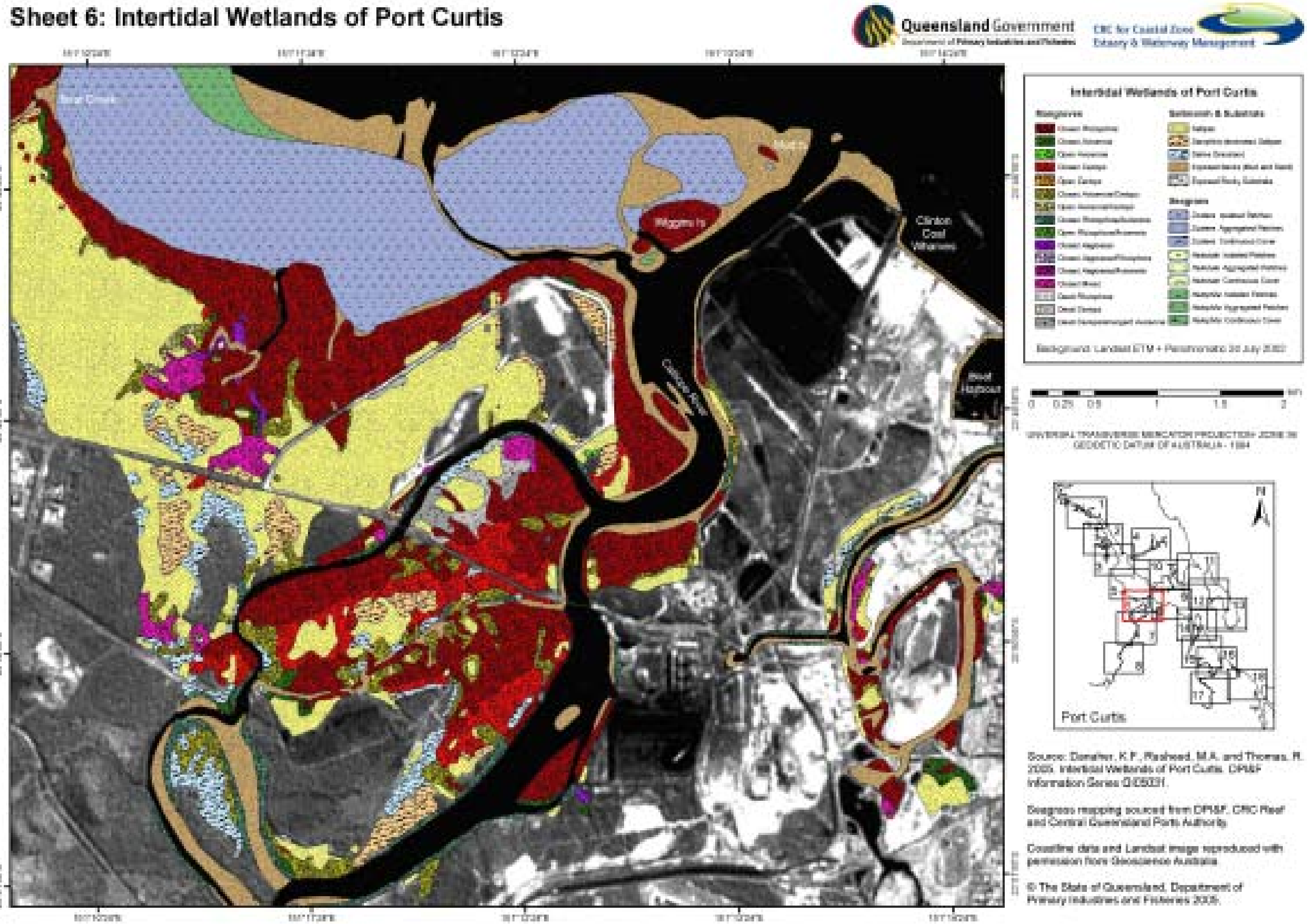

Figure 2.4. An example of one of the 18 maps showing the distribution of mangroves, saltmarsh, substrate and seagrass in Port Curtis

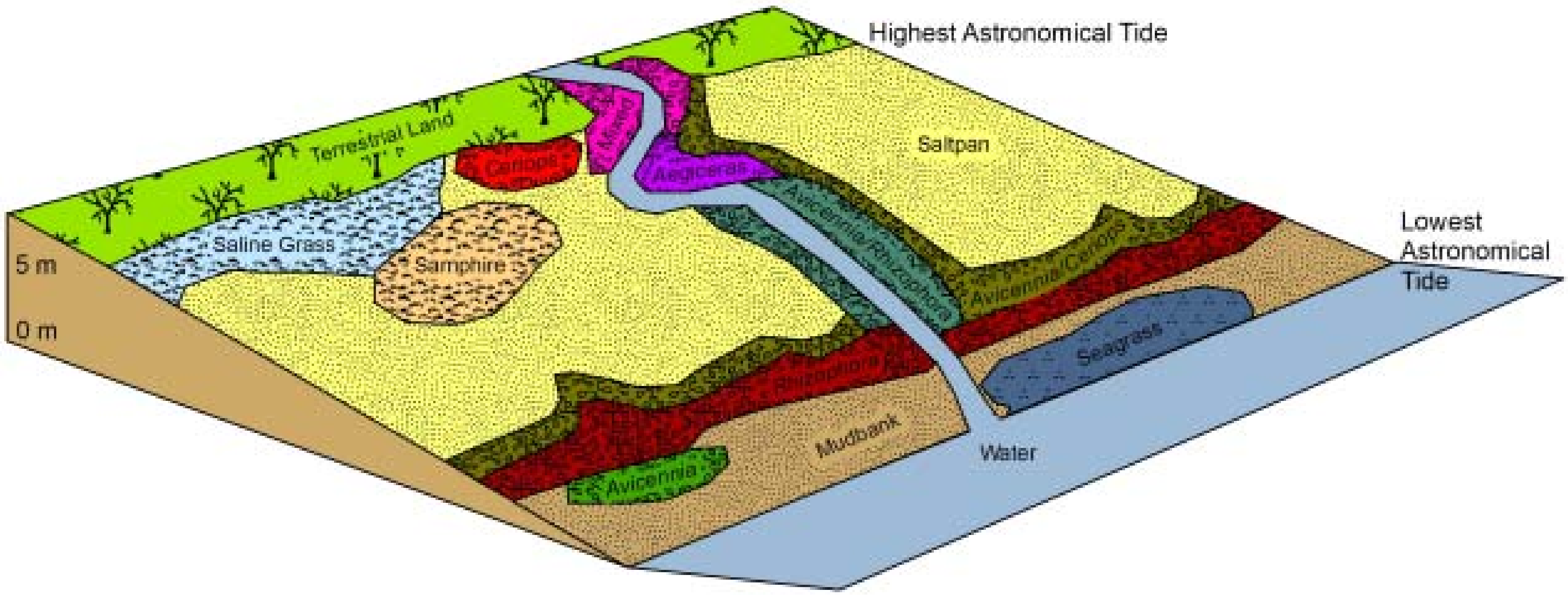

Figure 2.5. The general tidal profile of Port Curtis

Saline grass and samphire habitats together constitute saltmarsh, and can also occur immediately landward of mangroves.

2.4 Discussion

2.4.1 Regional significance of the intertidal wetlands of Port Curtis

Biogeographically, Port Curtis falls within the Shoalwater Coast bioregion as defined in the *Interim Marine and Coastal Regionalisation for Australia* (IMCRA Technical Group 1998). This bioregion includes the coastal and island waters from Mackay south to Baffle Creek. The inshore coastal region comprises large bays with very large tidal ranges (up to 6 m), large coastal islands, mostly sandy substrates, and little terrestrial input due to relatively (in comparison to neighbouring areas) low rainfall (1020 mm annual average at Gladstone). Figure 2.6 compares Queensland’s coastal wetland types by bioregion (De Vries *et al.* 2002). Apart from Wellesley-Karumba, the Shoalwater Coast has the largest proportion of saltmarsh/saltflats. This is probably due to the relatively low, highly seasonal rainfall with high evaporation rates (Saenger 1996). *Rhizophora* is also the dominant mangrove community type, which is typical throughout Queensland. South of Fraser Island *Rhizophora* is replaced by *Avicennia* as the dominant mangrove community type in southern Australia. *Avicennia* is also the dominant mangrove community type in the southern Gulf of Carpentaria.

The total number of mangrove species reported for the Curtis Coast study area is fourteen. Three species (*Acanthus ilicifolius*, *Bruguiera exaristata* and *Xylocarpus moluccensis*) are at their southern limits in the Curtis Coast region. The most southern *Xylocarpus moluccensis* on the eastern Australian coast occur near Clinton Bridge on Auckland Creek. *Xylocarpus granatum* occurs in the Fitzroy River delta to the north of Port Curtis but is then absent until Fraser Island and the Mary River (Saenger 1996). This discontinuity is unusual for mangrove distributions on the east coast of Queensland, but may be related more to freshwater availability than to temperature. *Bruguiera* spp. were observed only in Graham Creek but are plentiful both to the north (Bruinsma 2000) and south of Port Curtis (Bruinsma & Danaher 1999).

Saenger (1996) conducted one of the longest continuous mangrove studies in Australia, observing mangrove demographics in Port Curtis from 1974–1983. Indications are that the watercourses are currently infilling while the region is becoming somewhat more arid. The mangroves reflect this by displaying highly seasonal patterns of growth and deciduousness and highly seasonal leaf litter production, and by the fact that species which are at their southern limits are species generally found in more humid areas.

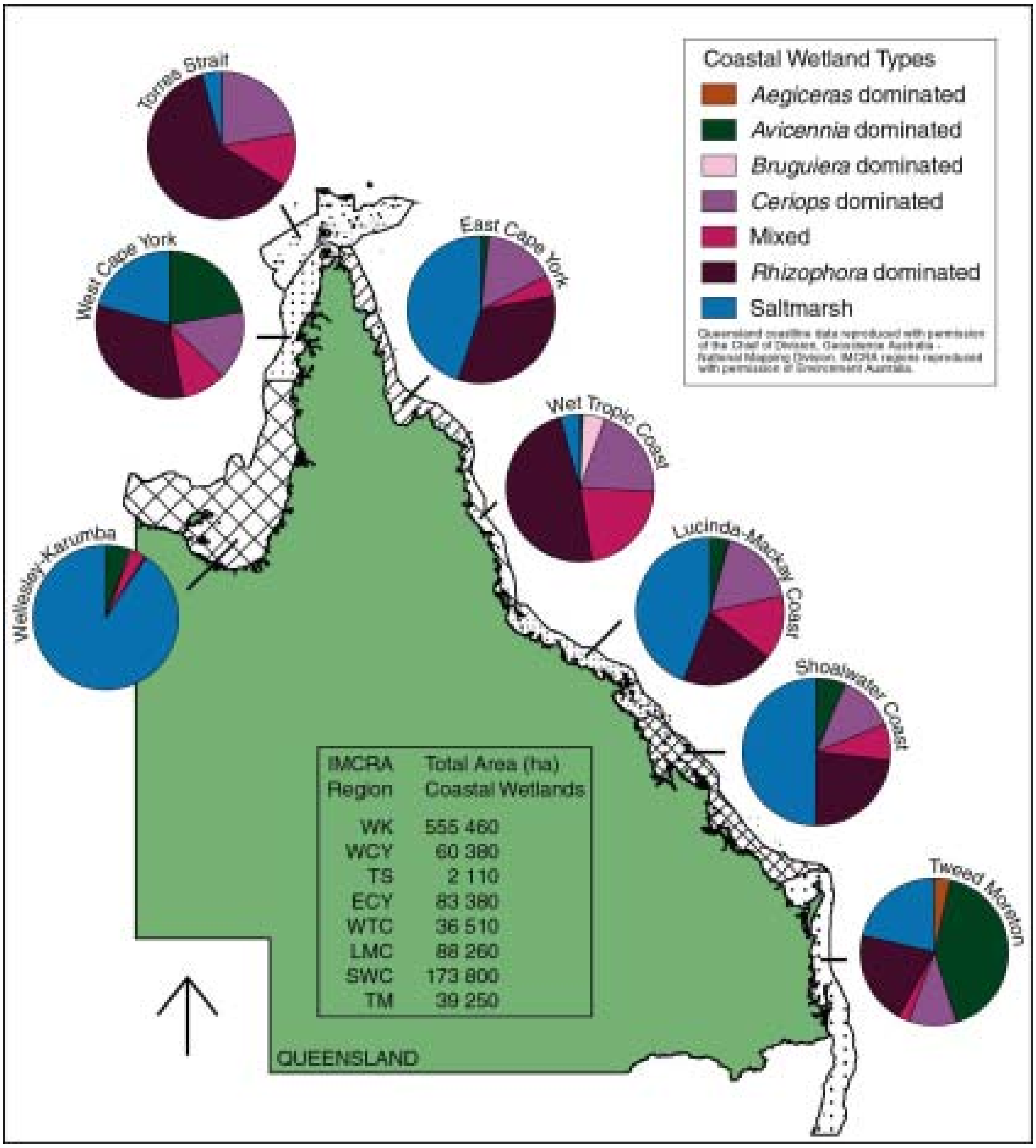

Figure 2.6. Queensland's coastal wetlands by IMCRA region
(IMCRA = Interim Marine and Coastal Regionalisation for Australia)

The extensive areas of coastal seagrass in Port Curtis are the only described large areas of seagrass for 170 km to the north and south, between Hervey Bay and Shoalwater Bay (Coles *et al.* 1990; Rasheed *et al*. 2003), giving them significant regional importance.

2.4.2 Current condition of the intertidal wetlands

Changes in the spatial area of mangroves and saltmarshes in Port Curtis have been well documented by Arnold (1996) and Duke *et al*. (2004). Duke *et al*. (2004) estimate that over 1600 hectares of tidal wetlands were lost between 1941 and 1999. Most of this loss was around Gladstone (particularly Auckland Inlet, and Calliope and Boyne River mouths) with large-scale reclamation for industrialisation, urbanisation and port development. There is still more expansion planned for the Port Curtis region, both industrial and residential.

Most of the mangrove and saltmarsh communities observed in this study were in good condition. Some of the saltmarshes on Curtis Island have been extensively damaged by pigs (Chris Hall, EPA pers. comm.). Unfortunately, the scale of this study was not detailed enough to detect this. Many mangrove communities were in near pristine condition. However, this study did find two locations that had severely stressed communities with dead mangroves. The first was a large area on the anabranch of the Calliope River. Houston (1999) suggested that this was a result of a severe hail storm which occurred during October 1994. Houston (1999) studied the types of damage sustained and found them to include: leaf damage and removal; bark shredding on one side of stems; and extensive small branch breakage. Recovery following the storm has been minor and eleven years later, communities still show signs of the impact. This study identified a community of *Ceriops* that are dead trees still standing. A community of dead *Ceriops* with emergent live *Avicennia* was also observed in the same location. After the hail storm, Houston (1999) found that *Avicennia* had a notable recovery, while *Ceriops* and *Rhizophora* died. Over 200 hectares of mangroves and saltmarsh were affected by the hail storm.

This study also identified from high resolution aerial photography a community of dead *Rhizophora* in Shoal Bay on Facing Island. This was not field-checked and possible causes were not investigated.

The 2002 baseline study of seagrass in Port Curtis by Rasheed *et al*. (2003) found extensive seagrass growing in the intertidal areas. Many of these communities occurred in proximity to port infrastructure, developments and dredged channels. However, subtidal seagrass communities that occur in other Queensland locations with similar or greater turbidity, such as Mourilyan Harbour (Thomas & Rasheed 2004), Cairns Harbour (Campbell *et al*. 2003) and Upstart Bay (Rasheed & Thomas 2002) did not occur in the inner harbour of Port Curtis. This is likely to be due to strong currents in the relatively narrow channels and rapid drop-off from the banks in the inner port.

Seagrass meadow area and biomass are likely to vary seasonally and between years. Generally, seagrass meadows on the east coast of Queensland peak I biomass and area in later spring/summer and are lowest over winter (e.g. McKenzie 1994; McKenzie *et al*. 1998; Rasheed 1999). It is likely therefore that the distribution and abundance (biomass) recorded in the baseline represented the peak for 2002. For other ports where seagrass has been monitored over several years such as Mourilyan, Cairns, Karumba and Weipa (Thomas & Rasheed 2004), 2002 was a year of below-average seagrass biomass and area for many of the meadows that have been monitored. It is possible therefore that seagrass biomass and area for Port Curtis may have been higher in other years than that measured in the 2002 baseline. Future monitoring is planned to occur during the same season as the 2002 baseline.

Rasheed *et al*. (2003) conclude that the presence of apparently healthy seagrass meadows in such close proximity to port facilities indicates that the port marine environment is relatively healthy. However, due to their location, these seagrass meadows would be vulnerable to direct impacts by future port infrastructure developments such as wharves, breakwaters and reclamation.

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Chapter 3. Assessment of sediment chemistry and relationships with intertidal wetland habitats in Port Curtis

Peter R. Teasdale, Mark A. Jordan, Ryan J.K. Dunn and David T. Welsh

3.1 Introduction

Sheltered coastal zones and soft sediments provide important habitat for infauna, epibenthic and pelagic organisms. The sediment also plays an important role in ecosystem processes and element cycling (Roubaud *et al*. 1996, Riedel *et al*. 1997). Fine, soft sediments generally retain large amounts of water at low tide and act as a buffer at high tide against changes in salinity, temperature and pH that may occur in the overlying water (Little 2000). Assuming there is a local supply of sediment, a combination of hydrological factors such as waves, tides and currents and vegetation type are responsible for determining where sediment is deposited. The volume and type of sediment deposited affects (and is affected by) the growth of vegetation on the shoreline (Little 2000). The following subsections provide information about intertidal habitats, additional to Chapter 1, focusing on relationships with sediment.

3.1.1 Intertidal muddy shores

Plants do not occur on all soft shores; their distribution is determined by factors such as: i) shelter; ii) slope; and iii) sediment supply (Little 2000; Roy *et al*. 2001). Unvegetated muddy shorelines are most common in tropical Australia where rainfall levels are very high and rivers carry a lot of water (Cronk & Fennessy 2001). This flushes large quantities of fine sediment and organic material into coastal areas, where they can form extensive muddy shores (mudbanks). The physical properties of sedimentary shores (i.e. sandy or muddy) are largely determined by exposure to water movement and the tidal range (Roy *et al*. 2001). In estuaries there is often a gradient from shores in exposed areas (usually near the mouth), which are comprised mostly of coarse sediments, to more sheltered shores, which have a larger proportion of fine sediments (Cronk & Fennessy 2001). The relative proportions of sands, silts and clays in sediment affect their chemical properties. In muds, the interstices are small and water exchanges slowly between the sediment and the overlying water column (Cronk & Fennessy 2001); the reverse is true in sandy sediments.

3.1.2 Seagrasses

Seagrasses typically occur in shallow water that overlies soft sediment, from cool temperate to tropical regions (Mann 2000). Generally, the distribution of seagrasses is dependent upon factors affecting photosynthesis, settlement, recruitment and desiccation. In tropical locations, seagrasses normally do not occur on mudbanks that are exposed at low tide, due to desiccation stress (Wolanski *et al*. 1997). If the sediment is suitable, seagrasses will extend to a depth where the annual light flux is just sufficient to support a positive balance of photosynthesis over respiration (Mann 2000). Assuming salinity and temperature are not limiting, factors affecting seagrass growth include light attenuation features, sediment characteristics and nutrient conditions (Livingstone *et al.*1998; Roy *et al*. 2001). Low nutrient levels will favour the growth of seagrasses, as they are able to fix a large proportion of their nitrogen requirements via symbiotic bacteria, whereas higher nutrient concentrations may enable algae to outcompete and shade seagrasses and can therefore lead to decreases in seagrass cover (Welsh 2000). The mean rate of water movement may also affect seagrass distribution. If water movement is too strong, colonisation and establishment may become difficult and if water movement is not sufficient, uptake of carbon dioxide and nutrients may result in their depletion within the boundary layer of the leaves (Mann 2000).

Reductions in seagrass cover/range are not unusual and can be caused by both anthropogenic (Onuf 1994) and natural processes. Decreased light availability has been recognised as the major cause of seagrass declines (Onuf 1994; Hall *et al*. 1999). Chronic light reduction may be a direct consequence of increased suspended sediment loads from flooding (Campbell & McKenzie 2004), dredging (Onuf 1994) or secondary effects from increased runoff following terrestrial clearing or persistent microalgal blooms attributable to increased nutrient loads (Duarte 1995). Alternatively, seagrass dieback is not caused by chronic light reduction and its causes are still poorly understood (Hall *et al*. 1999).

In a local study, Campbell and McKenzie (2004) monitored the reestablishment of *Zostera capricorni* following its large-scale range contraction from flooding. They found that pre-flood seagrass cover was restored after 24 months, despite having not germinated for 18 months. This indicates just how quickly the species can recover once established and secondly that seeds of this species may remain viable for several years. Seedling growth and therefore recolonisation of *Z. capricorni* is dependent upon both seasonal (temperature and day-length) and physicochemical parameters of the sediment and overlying water. Germination is favoured with low salinities, cool water temperatures and anaerobic sediments rich in organic matter (Brenchley & Probert 1998).

Seagrass beds are found scattered throughout Port Curtis, on both mud and sandbanks. Regular monitoring is undertaken on a relatively large bed situated at the mouth of the Calliope River (Chapter 2). The dominant seagrass communities in the area consist mostly of *Zostera capricorni* and *Halophila ovalis*, with *Z. capricorni* dominant. Another local study found that *Z. capricorni* does not grow in strong currents and like all seagrasses is light limited, but grows better with longer immersion times (Young & Kirkman 1975). Young and Kirkman also noted that in one area of their study site a massive mortality of *Z. capricorni* rapidly occurred, which is thought to be caused by a disease followed by extensive grazing. The loss of the dominant *Z. capricorni* was followed by the loss of *H. ovalis* as the roots could not withstand the hydrological stress without protection from the dominant seagrass (Young & Kirkman 1975).

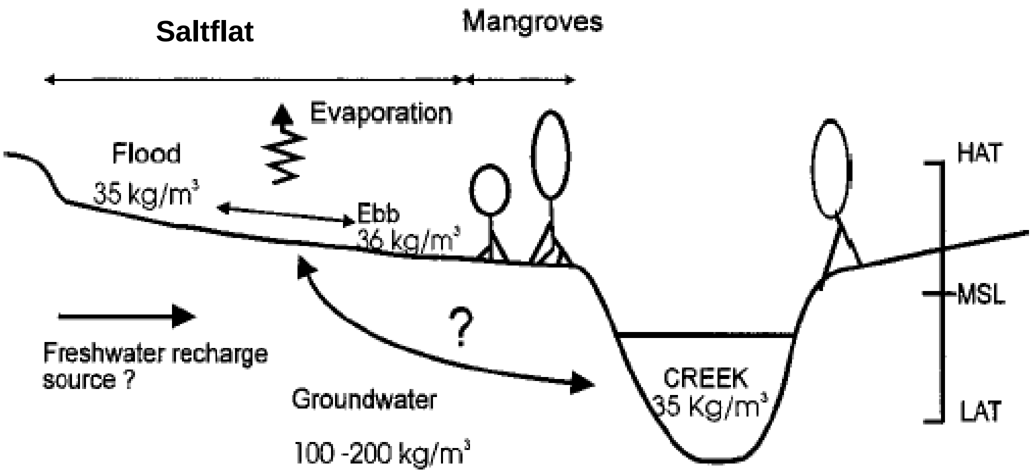
3.1.3 Saltflat

Saltmarshes include saltflats, samphire and saline grasslands. Saltflats are typically formed from bare depressions within saltmarshes that gradually retard plant growth due to continual water logging and increased salinity (Packham & Willis 1997). Vegetated saltmarshes are quite common in temperate Australia, but saltflats predominate in the tropics (Underwood & Chapman 2000). In the dry tropics, saltflat habitat is a large source of nutrients to the near-shore zone and often occupies ten times more area than that occupied by mangroves, yet saltflat habitat is probably the least studied of all intertidal communities (Ridd *et al*. 1997).

Saltflats occupy the highest elevation within the intertidal zone. They are characterised by hypersaline groundwater and high evaporation rates (Ridd *et al*. 1997). A clear distinction normally exists between mangrove and saltflat habitat, and mangroves on this boundary are often stunted due to the hypersaline conditions (Kennish 1986). Saltflats are extremely level, typically with longitudinal surface slopes of around 10^{-4} (Ridd *et al*. 1997). In general, saltflat surface sediment is relatively unconsolidated clay and below approximately 15 cm is highly cohesive clay with a very low permeability (Ridd *et al*. 1997). Very low permeability equates to very slow transport of groundwater through the sediment, which will affect salt and nutrient fluxes.

Saltflats may not be covered by seawater for a period of a fortnight to several months, depending on the tidal curve (Ridd *et al*. 1997). During this period porewater salt concentrations increase greatly, sometimes as high as 200 ppt

(Ridd & Sam 1996) due to evaporation (Ridd *et al*. 1997). Seepage of groundwater has the potential to deplete the sediment of a considerable amount of salt (Ridd *et al*. 1997); however this is completely dependent upon the permeability, which is typically poor. The processes maintaining a typical saltflat/mangrove environment are illustrated in Figure 3.1. Typically, the zone immediately above the highest astronomical tide (HAT) mark will consist of freshwater coastal vegetation such as *Melaleuca* and *Casuarina* species. The elevation difference between the mangroves and the creek will generally be far less exaggerated (Figure 3.1) and will often consist of extensive mudbank habitat, as was the case in this study.



HAT = highest astronomical tide; LAT = lowest astronomical tide; MSL = mean sea level; kg/m^3 = ppt

Figure 3.1. Illustration of a characteristic saltflat/mangrove cross-section, showing possible water flow pathways and typical salt concentrations (Ridd *et al*. 1997)

In this study, the upper intertidal zone behind the mangroves was found to be dominated by saltflats in many areas. The processes that form and sustain these saltflats typify those found throughout the dry tropics. Small patches of saline grasslands and samphire were scattered throughout saltflat habitat in Port Curtis. Saline grassland habitat patches were more commonly found along or nearby saltwater drainage creeks or sources of fresh water.

3.1.4 Mangroves

Mangroves occur mostly where shores are sheltered, that is, behind shingle bars, in lagoons and in estuaries. Mangroves develop best on gently sloping shores that are not excessively supplied with sediment (Little 2000). Sediment accumulation in these areas facilitates seedling development, which fosters community expansion, particularly in areas with large tidal ranges (Kennish 1986). In turn, mangrove roots help to stabilise and trap additional fine sediment (Underwood & Chapman 2000). Mangroves are most common and diverse where mean temperatures in the coldest months exceed 20°C and do not occur where mean monthly temperatures fall below 10°C, where saltmarshes dominate.

Mature mangrove communities are dominated by branching patterns of drainage creeks with slightly elevated vegetation areas between and sometimes with isolated lagoons up to the mean high water neap mark (MHWN) (Little 2000). Mangrove growth is more vigorous along creek margins, probably due to nutrient-rich overbank deposits and better drained soils (Carter 1988). Water flow through mangroves is also thought to be important in distributing nutrients (Carter 1988).

Where growing conditions are favourable, porewater salinity and water content are thought to be the major parameters influencing mangrove habitat zonation. The porewater salinity of sediments is a function of local precipitation, subterranean seepage, terrestrial runoff, evaporation and tidal flushing (Kennish 1986). Due to their slower growth rates, mangroves are outcompeted by freshwater plants in the upper end of estuaries (Little 2000). In tropical areas with relatively low rainfall (e.g. Port Curtis), the upper intertidal distribution of mangroves abruptly gives way to saltflat habitat and, in more temperate climates, saltmarsh (Ridd & Sam 1996).

Ridd and Sam (1996) analysed porewater salt concentration gradients across mangrove and saltflat transects in Townsville, Queensland, very similar to the Post Curtis site. There was a distinct rise in porewater salt concentrations corresponding with the change in habitat from mangrove to saltflat (Figure 3.2). They found no difference in elevation or in sediment characteristics at this interface and therefore they concluded that some mechanism is reducing the groundwater salt concentration in the mangrove habitat. They proposed three possible explanations including i) mangrove sediments may have a lower rate of evaporation due to shading; ii) mangrove habitat may have a more efficient exchange mechanism to transfer pore water to the overlying water at high tide (via animal burrows for example); and iii) biological mechanisms of salt extraction in some mangrove species may significantly reduce the sediment porewater salt concentration (Ridd & Sam 1996).

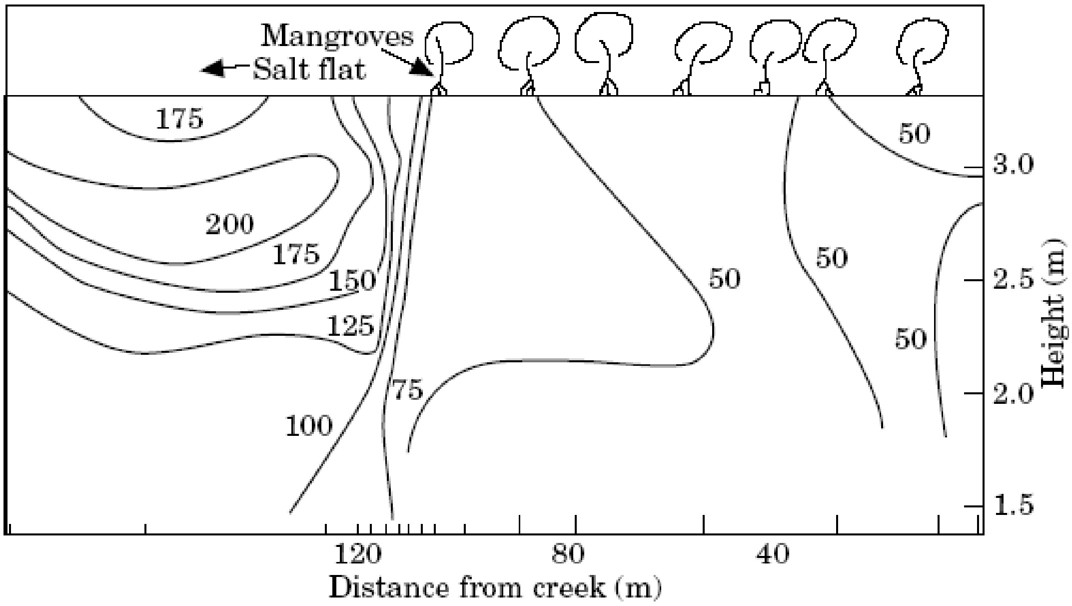


Figure 3.2. Porewater salt concentration (ppt) gradient along a mangrove/saltflat transect in Townsville, Queensland. Heights relative to lowest astronomic tide (Ridd & Sam 1996)

In temperate climates, Harty and Cheng (2003) found that saltmarsh vegetation was more salt-tolerant than mangroves and their distribution was dependent upon the sediment water content. Infrequent tidal inundation, and therefore low water content and high porewater salinity, is necessary to maintain the saltmarsh community and discourage encroachment by mangroves.

It has been suggested that mangroves may be more prevalent in tropical and subtropical climates because it may be easier to cope with salt stress at higher temperatures (Little 2000). Other halophytes and freshwater plants in particular are outcompeted in the mangrove environment due to salinity stress, high sulfide concentrations in the soil, root zone anoxia, low light due to shading and crab predation of seeds and propagules (Cronk & Fennessy 2001).

Mangroves show distinct zonation patterns, with individual species dominating different zones depending upon the abovementioned physical, biological and chemical parameters. Other factors which limit mangrove distribution and growth of a given species include: mean rainfall (the source of groundwater), topography (which directs groundwater flow), dispersibility of mangrove propagules, tidal influence and coastal geomorphology, which influences sediment types (Tomlinson 1986). Mangrove patches tend to be influenced by three main factors over time: i) recurrence rate of individuals following disturbances; ii) changes in hydrology; and iii) frequency of insect and disease outbreaks (Schaeffer-Novelli *et al*. 2000). Recurrence rate following natural and anthropogenic disturbances may be important, as Port Curtis is in an area that may be prone to fire and severe storm disturbances.

3.1.5 Objectives

This study compared sediment physicochemical measurements between four intertidal habitats (mudbanks, mangroves, saltflat and seagrass) found within a subtropical estuarine system. The primary objective of this work was to investigate differences and relationships between sediment and porewater characteristics that may provide insights into intertidal habitat zonation. Put simply, researchers sought to determine what are the principle sediment parameters that promote or discourage a given habitat type.

3.2 Methods

3.2.1 Study area and experimental design

Sediment sampling was conducted at two intertidal locations in Port Curtis, Queensland (Kangaroo Island and south of Fisherman's Landing) during February 2004 (Figure 3.3). Seagrass sites near the Calliope River, sampled as part of the follow-up study are also shown in Figure 3.3. Four additional seagrass sites were at South Trees Point (not shown on map). It should be noted that the vegetation map shown in Figure 3.3 predates this study and therefore may differ from the vegetation maps in Chapter 2, but is included because a vegetation map similar to this was used to choose the sampling sites.

Sampling was conducted along transects and at randomly determined locations. The transects were carefully chosen to span large patches of different habitat types (mudbank, mangrove and saltflat). The transects were 2–3 km long with site intervals not less than 50 m (average ~200 m). Random sampling was conducted for each of these habitat types at both locations. Saline grassland habitat was also randomly sampled at both locations. Random sampling was primarily undertaken to provide additional data that may be used in conjunction with transect measurements to evaluate the mean conditions for each habitat for a given analyte. Due to bad weather during the first sampling trip, random sampling of seagrass habitat was delayed until August 2005. Seagrass sediment cores were sampled from near the Boyne smelter wharf (north of the South Trees Inlet—sites 5–8) and from the routinely monitored seagrass beds at the mouth of the Calliope River (sites 1–4). All of the sample sites are shown in Figure 3.3.

It is important to note that there was no evidence of seagrasses (i.e. root mass, leaf material or seeds) found in any of the cores, as would be expected if the beds were colonised with seagrasses at the time. This phenomenon is not unusual as seagrass beds typically decrease and increase in area over time. Consequently many of the ‘seagrass’ results may not reflect sediment conditions when substantial seagrass is present. The experimental design is summarised in Table 3.1.

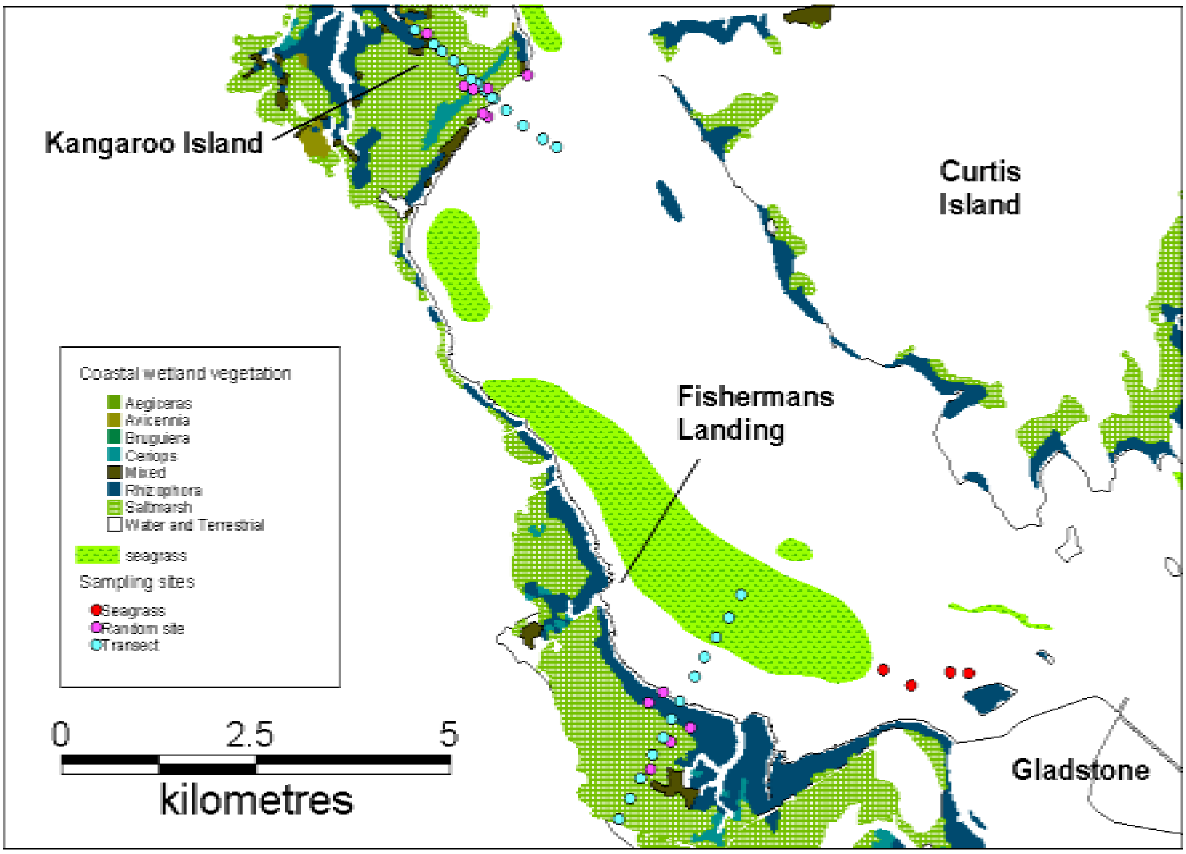


Figure 3.3. Transect and random sampling sites for each intertidal ecosystem at Kangaroo Island and near Fisherman’s Landing

Table 3.1. Experimental design and number of sites sampled at each location

Habitat type	Kangaroo Island	Fisherman’s Landing	Seagrass sites
Transect site #	**15**	**12**	
Mudbanks	5	5	
Mangroves	5	2	
Saltflats	4	4	
Saline grassland	1	1	
Random site #	**8**	**5**	**8**
Mudbanks	4	–	
Mangroves	2	3	
Saltflats	1	1	
Saline grassland	1	1	
Seagrass			8
Total measurements (Total sites x 3 depths)	**69**	**51**	**24**

3.2.2 Sampling methods

Sediment samples were collected using 400 mm long x 44 mm diameter PVC hand corers. Two cores were collected at each site, extruded and homogenised on site or placed on ice intact and extruded later that day in the laboratory. The top 0–3 cm from each core was analysed and two other 3 cm sections from each core were randomly chosen to determine average measurements for the 0–40 cm depth profile.

Separate subsamples of the sediment were immediately taken following homogenisation, for later analysis of exchangeable ammonium (NH_4^+) and exchangeable phosphate (PO_4^{3-}). A summary of all measurements and analytes, including preservation techniques, is displayed in Table 3.2.

Table 3.2. Chosen analytes, measurements and preservation techniques

Analyte/measurement	Preservation/storage
Particle size analysis	Frozen
Porosity	Frozen
Total organic carbon (TOC)	Frozen
Porewater salinity	Frozen
Acid volatile sulfides (AVS)	Frozen
Total Kjeldahl nitrogen (TKN)	Frozen
Exchangeable ammonium (NH_4^+)	1M KCl preservation, frozen
Total phosphorous (TP)	Frozen
Exchangeable phosphate (PO_4^{3-})	1M $MgCl_2$ preservation, frozen

All analytes were prepared, digested and measured using standard or published techniques as much as practical. These procedures are outlined in Table 3.3.

Table 3.3. Methods used for each analyte, references, recoveries and limits of detection (LOD)[1] where applicable

Analyte	Method	LOD (μg/g dry wt.)	Recovery (%)	Modified from:
TOC	Fixed and volatile solids ignited at 550°C	NA		APHA (2540E), 1999
AVS	Purge and trap	0.002[2] (n=4)	82.0 ± 7.4 (n=18)	Allen *et al*. 1993
TKN	Macro-Kjeldahl method	5 ± 5[3]		APHA (4500-N_{org} B), 1999
NH_4^+	Phenate method	0.050 (n=6)		APHA (4500-NH_3 F), 1999
TP	Stannous chloride method	12.8 (n=4)		Pardo *et al*. 2004; APHA (4500-P D), 1999
Exchangeable PO_4^{3-}	Stannous chloride method	0.270 (n=10)		APHA (4500-P D), 1999

3.3 Results and discussion

3.3.1 Porosity

Porosity measurements were found to generally decrease at both locations as habitats changed from mudbank – mangrove – saltflat. However, this trend was not as prominent for mean depth measurements at Fisherman's Landing. Porosity was found to increase again in saline grassland habitat (Figure 3.4). These trends were again evident when porosity measurements from both locations were combined and compared between habitat types (Figure 3.5).

[1] The method detection limit was determined by multiplying the standard deviation of the blanks by three. This measurement was then converted to a weight/weight concentration to calculate the limit of detection.
[2] Acid volatile sulfides LOD measured in μmoles/g dry weight.
[3] Quoted LOD by the NATA registered lab Queensland Health Scientific Services, Brisbane.

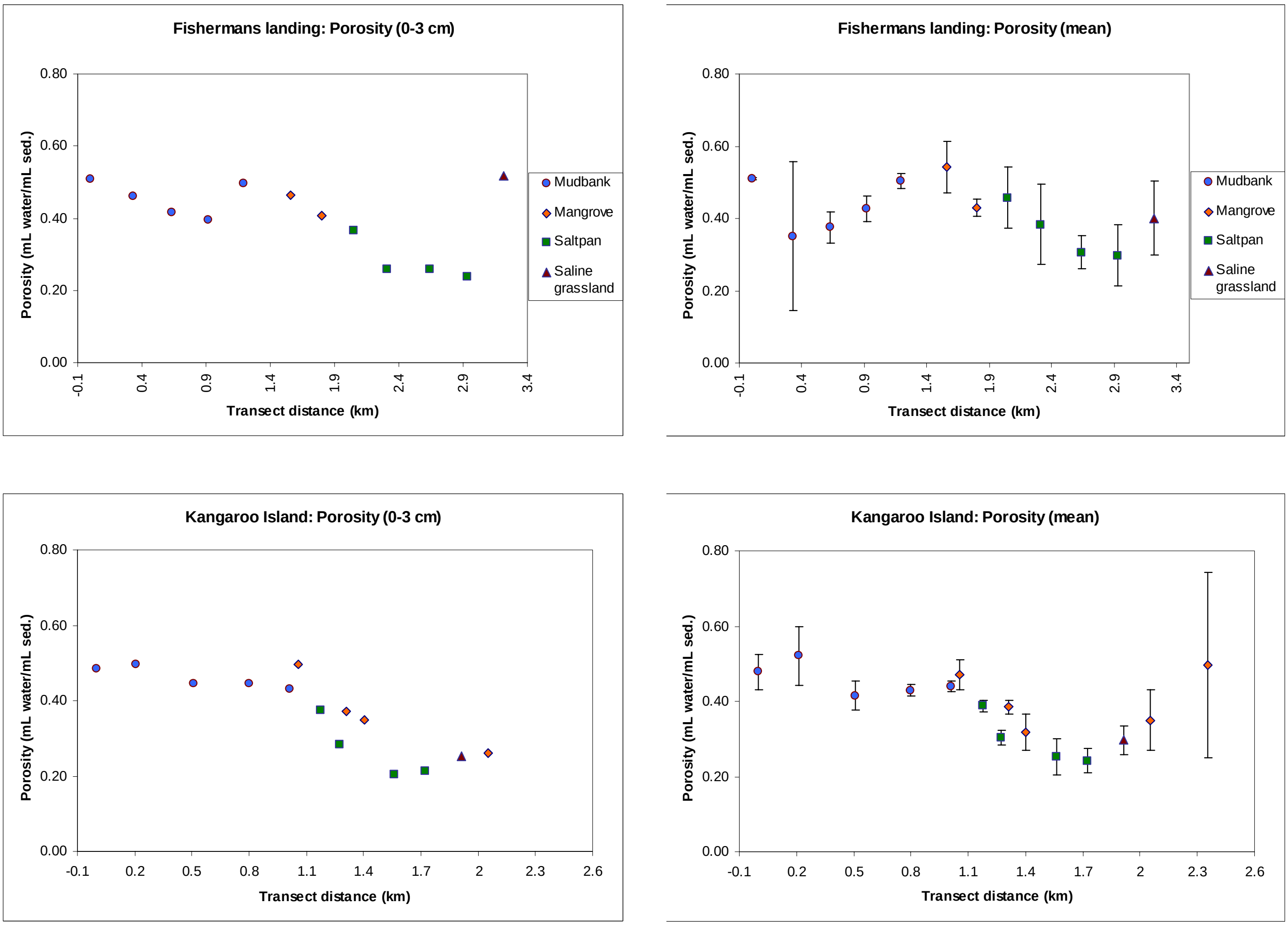


Figure 3.4. Transect porosity measurements (surface 0–3 cm and mean depth profile) at Fisherman's Landing and Kangaroo Island[4]

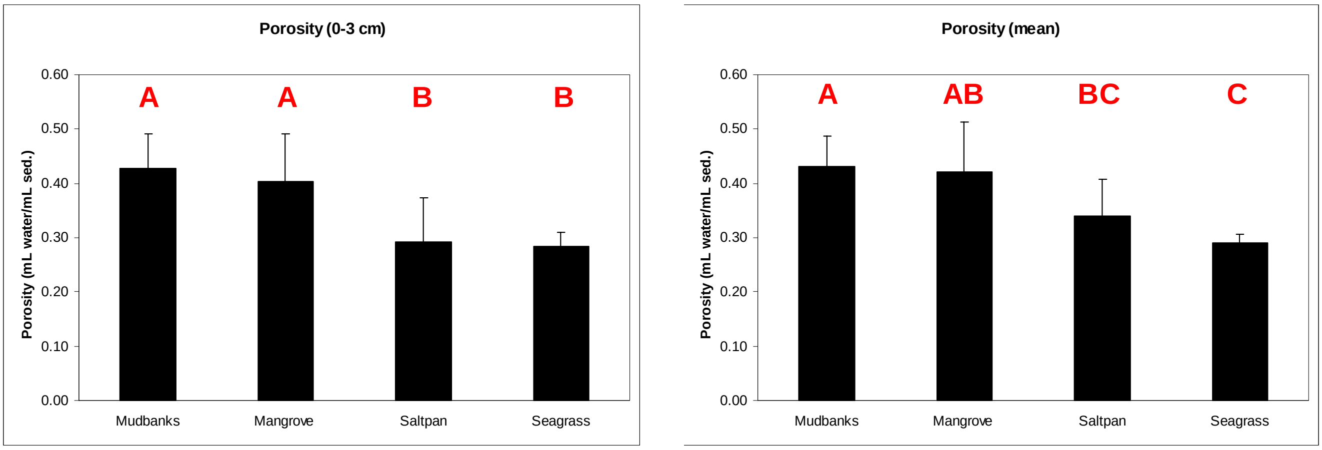


Figure 3.5. Mean (+1 s.d.) surface and mean depth profile porosity measurements from each habitat type [5]

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[4] Saltpan here and in all other figures is the same as saltflat as discussed in the introduction and throughout this chapter.

[5] Saline grassland measurements were analysed statistically with the saltflat habitat for all analytes.

One-way ANOVA analysis revealed that porosity was significantly different between habitat types for both the mean surface (df = 46, F = 12.753, p <0.001) and mean depth (df = 47, F = 11.407, p <0.001) measurements. Tukey's *post hoc* analysis found that mean surface porosity measurements were significantly lower within saltflat and seagrass habitat than both mudbank and mangrove habitats. Non-parametric Tamhane analysis was necessary for comparison of mean depth porosity measurements. Seagrass habitat was found to be significantly lower than both mangrove and mudbank habitat and saltflat habitat recorded significantly lower mean depth porosity than mudbanks only.

Seagrass habitat porosity measurements are thought to be typical of the given conditions during the sampling period (i.e. devoid of seagrass). It is thought that the seagrass beds may have been bare of seagrass (or at least where cores were taken) for quite some time. Generally, very fine sediment settles in areas where seagrass occurs; consequently under such conditions relatively high porosity measurements would be expected. The similarity of the surface and mean depth measurements may suggest that all of the fine surface sediment has been eroded away following the disappearance of the seagrass. We recommend that a more detailed study of seagrass dynamics and sediment biogeochemistry within Port Curtis is carried out in order to determine a possible reason for these observations.

It is no surprise that saltflat habitat was also found to have relatively low porosity despite theoretically having low permeability. As the saltflats are only periodically inundated with estuarine water with the highest of spring tides, most of the time the pore water is evaporating, consequently lowering the porosity and increasing the porewater salinity (Section 3.3.2). A positive relationship between porosity and particle sizes should exist, as fine sediments have relatively more pore spaces and therefore can hold more water (i.e. increased porosity). This was not found to be the case in this study as saltflat habitat was found to contain the finest sediment (Section 3.3.4), yet reported relatively low porosity measurements as a consequence of evaporation, particularly in the surface layer. Additionally, due to the hypersaline pore water (Section 3.3.2), approximately 10–15% of the porewater volume consists of salt and not water, which inevitably lowers the porosity even further. Given the unusual relationship between porosity and particle size measurements in saltflat habitat, porosity was used in this study as a surrogate for particle size analysis when discussing relationships with other analytes.

3.3.2 Porewater salinity

Porewater salinity measurements at both locations were found to considerably increase as habitats changed from mudbank – mangrove – saltflat and decrease again as transects passed through saline grassland or mangrove habitats again. This salinity gradient was particularly evident moving landward along the Kangaroo Island transect in particular (Figure 3.6). At the farthest inland sites, it is thought that terrestrial freshwater runoff has diluted the porewater salinity concentration sufficiently to allow mangroves to once again take over from saltflat habitat, but not so much as to allow colonisation of less salt tolerant species such as casuarinas.

These trends were again evident when porewater salinity measurements from both locations were combined and compared between habitat types (Figure 3.7). One-way ANOVA analysis revealed that mean porewater salinity measurements were highly significantly different between habitat types for both the surface ($df = 44$, $F = 64.835$, $p < 0.001$) and mean depth ($df = 47$, $F = 49.027$, $p < 0.001$) measurements; however equal variances were not assumed. Non-parametric Tamhane's *post hoc* analysis found that porewater salinity was significantly higher within saltflat habitat than all other habitats for both measurements. Seagrass salinity was found to be significantly higher than mudbank salinity for both measurements, although these results are primarily attributed to differences in the water column salinity at the time of sampling.

Porewater salinity appears to be a major factor influencing mangrove habitat zonation in this study as an abrupt salinity interface between habitat types was apparent. It is also highly probable that the presence of mangroves effectively lowers the pore water salt concentration (Ridd & Sam 1996). Mangroves were not found to grow in sediment with porewater salinity concentrations higher than 110 ppt. Saline grasslands were found to be more salt-tolerant, with surface salinity measurements reported as high as 140 ppt. Saltflat mean depth pore water salinities were found to range between 81–188 ppt.

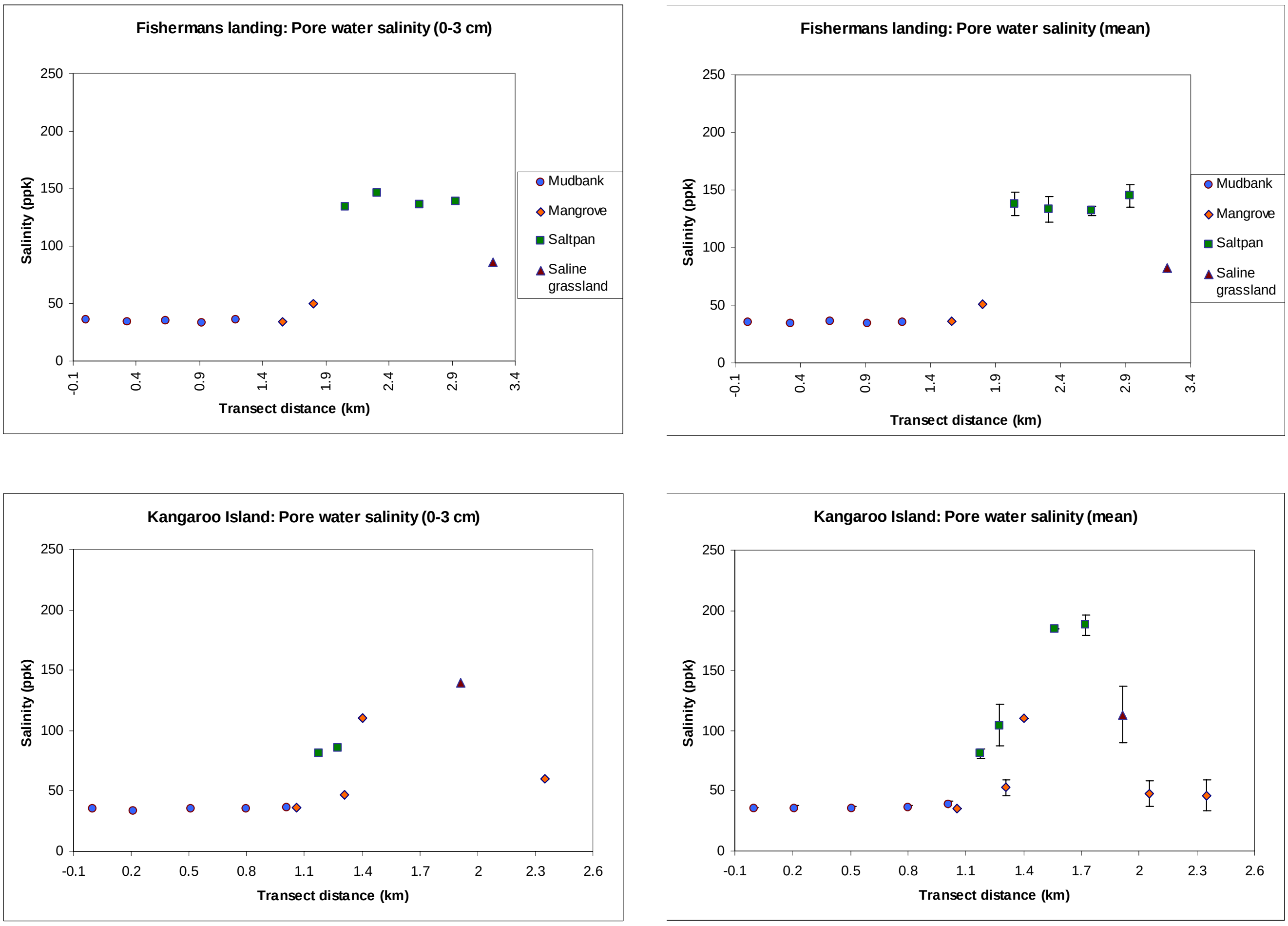


Figure 3.6. Transect porewater salinity measurements (surface 0–3 cm and mean depth profile) at Fisherman's Landing and Kangaroo Island

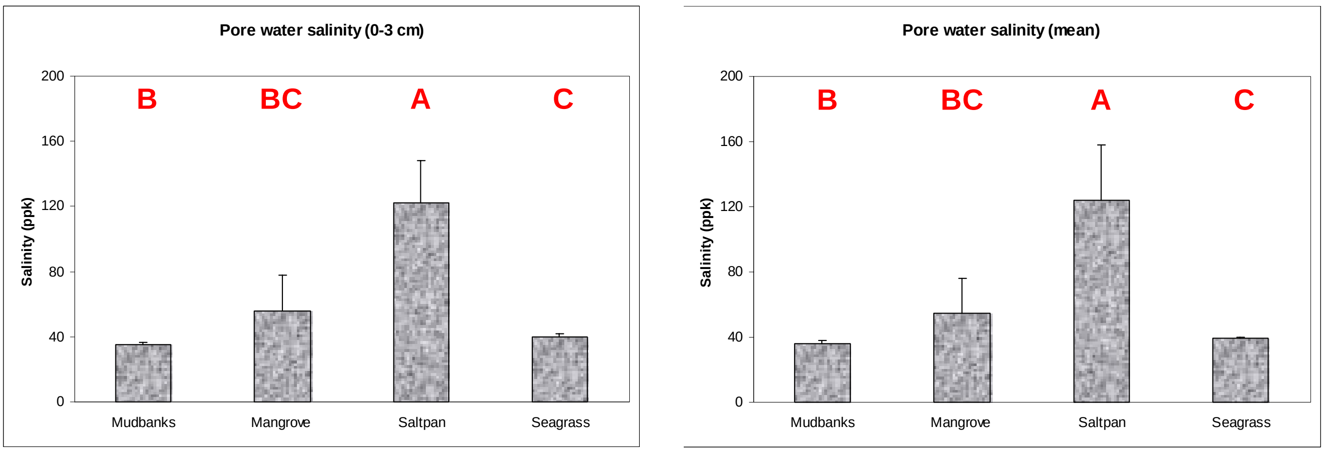


Figure 3.7. Mean (+1 s.d.) surface and mean depth profile porewater salinity measurements from each habitat type

3.3.3 Total organic carbon (TOC)

TOC measurements did not follow any discernable trends across habitats at either location and were relatively consistent. Mangrove TOC measurements appeared to be higher than mudbank measurements in particular (Figure 3.8). Mudbank and particularly seagrass habitats reported the lowest TOC measurements (Figure 3.9).

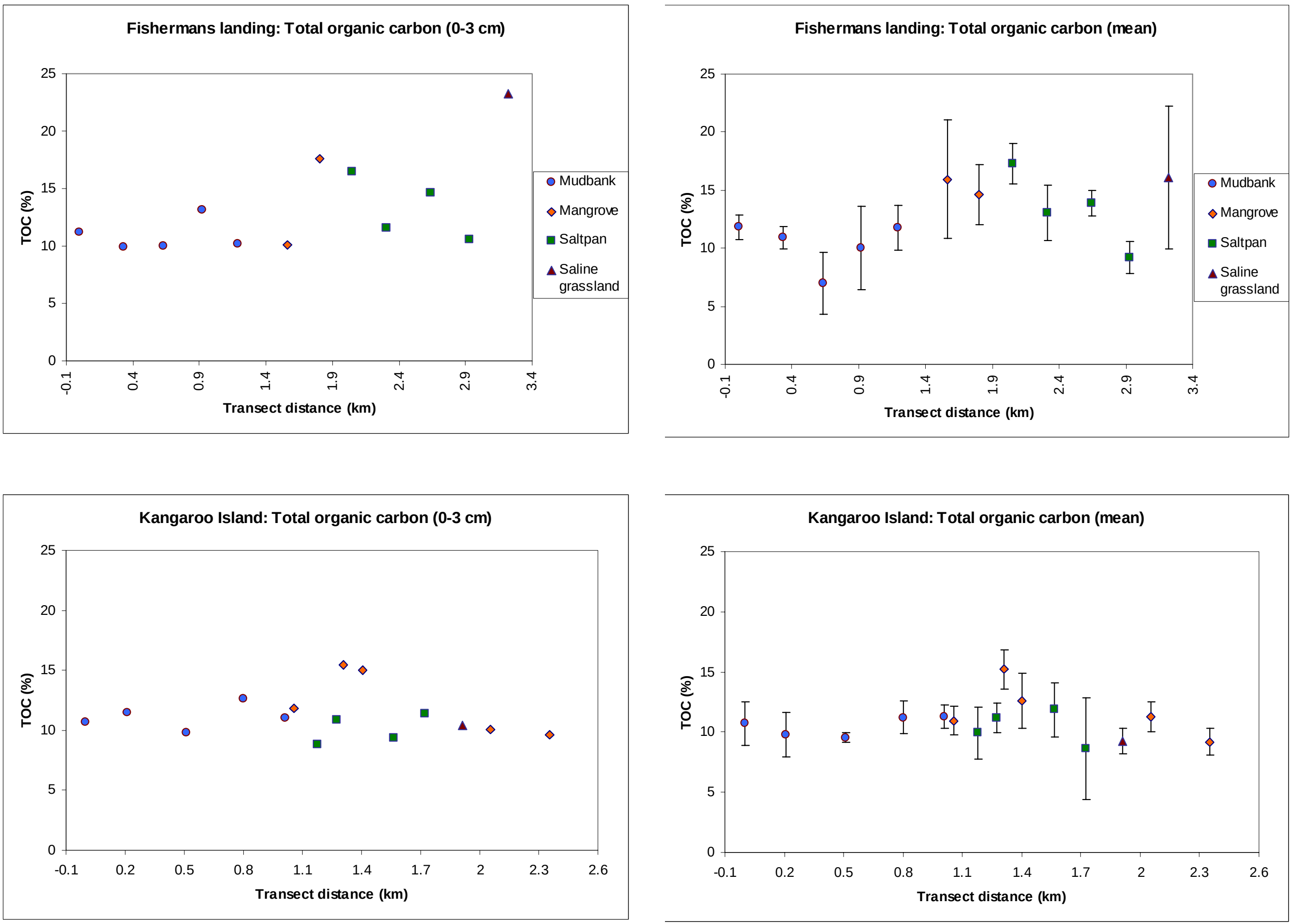


Figure 3.8. Transect total organic carbon measurements (surface 0–3 cm and mean depth profile) at Fisherman's Landing and Kangaroo Island

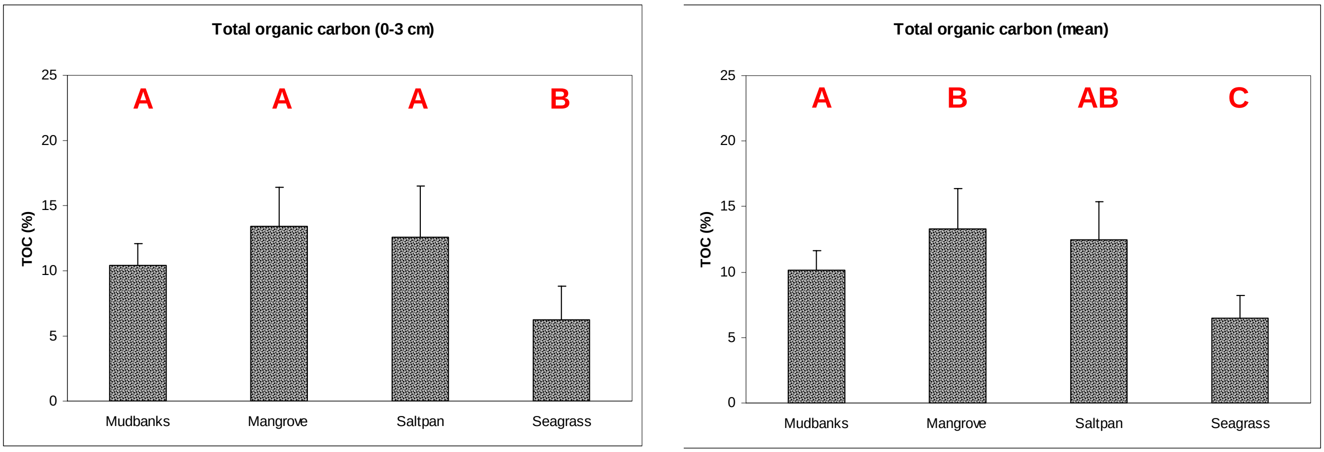


Figure 3.9. Mean (+1 s.d.) surface and mean depth profile total organic carbon measurements from each habitat type. Means with the same letter are not significantly different

One-way ANOVA analysis revealed that mean TOC measurements were significantly different between habitat types for both the surface (df = 47, F = 19.184, p <0.001) and mean depth (df = 47, F = 20.891, p <0.001) measurements, following natural log transformation. Tukey's *post hoc* analysis found that TOC was significantly lower within seagrass habitat than all other habitats for both measurements. Additionally, mean depth mangrove TOC measurements were found to be significantly higher than mudbank measurements, most probably due to the contribution of living and decomposing root biomass.

A general distinction can be made between TOC measurements from non-vegetated (mudbanks and seagrass[6]) and vegetated (all others) habitats. Vegetated habitats typically reported the highest TOC measurements, especially mangroves, as humus also contributes to the TOC pool. The net flux of organic carbon into saltflat habitat is thought to be lower than mangrove habitat; however it is thought that due to hypersaline pore waters (in saltflats), decomposition of organic carbon by micro-organisms may be considerably reduced (Ollivier *et al*. 1994), therefore leading to a gradual accumulation of carbon. During periods of seagrass coverage on seagrass beds, TOC measurements would be expected to be relatively high, possibly higher than mangrove habitat due to enhanced sedimentation of detritus and inputs of organic matter at depth as roots and rhizomes.

[6] Like saltflats, mudbank and seagrass habitat contains simple photosynthetic organisms such as diatoms and algal mats. However, it is thought that photosynthetic biomass would be relatively low, as light absorption decreases while the mudbanks and seagrasses are covered at high tide.

3.3.4 Particle size fractionation

The proportion of sediment <60 µm in size was found to generally increase along the transect at Kangaroo Island; however no discernible trends were evident at Fisherman's Landing for either the mean or surface depth measurements (Figure 3.10). This measurement was not carried out for the seagrass sites.

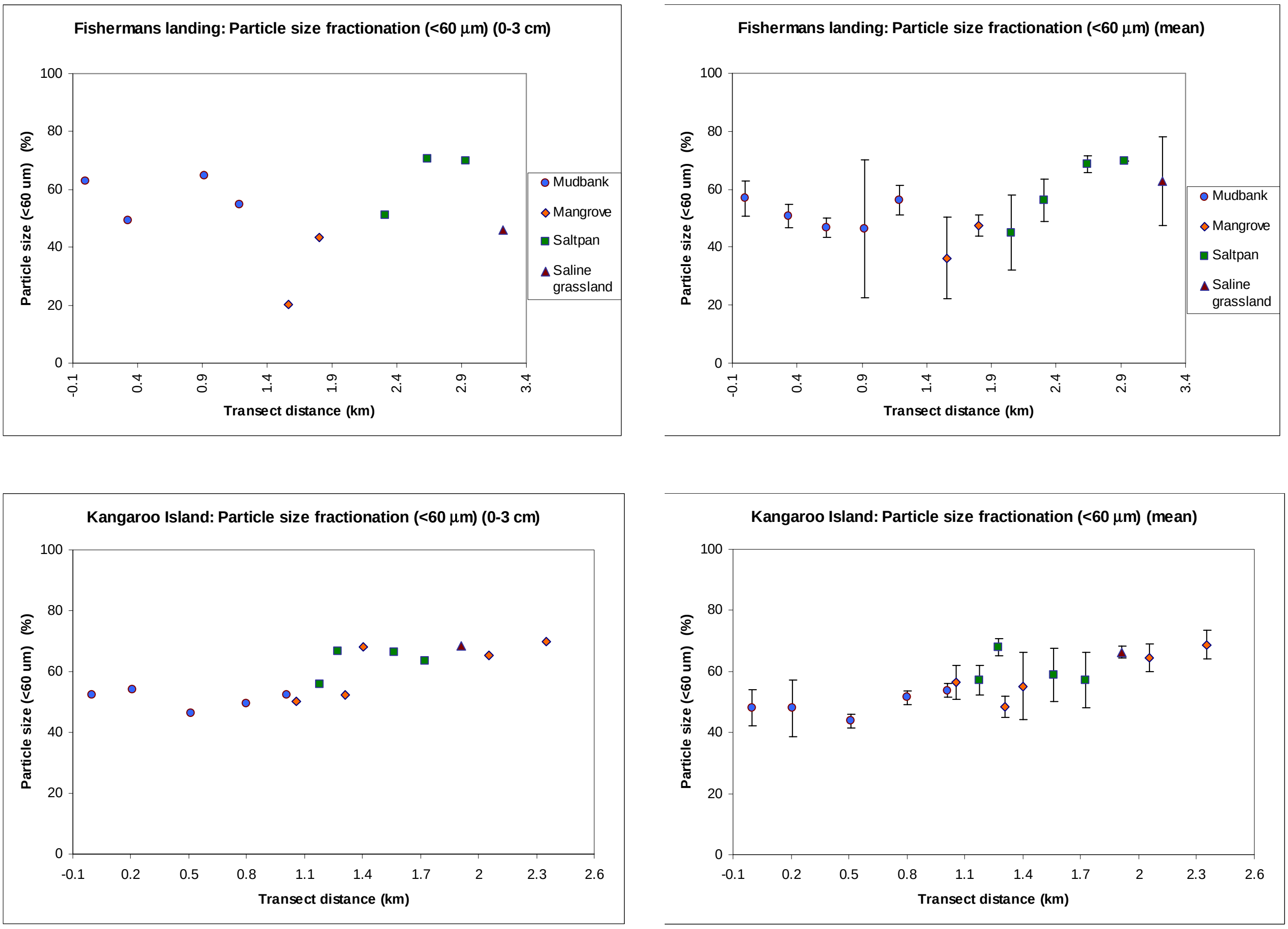
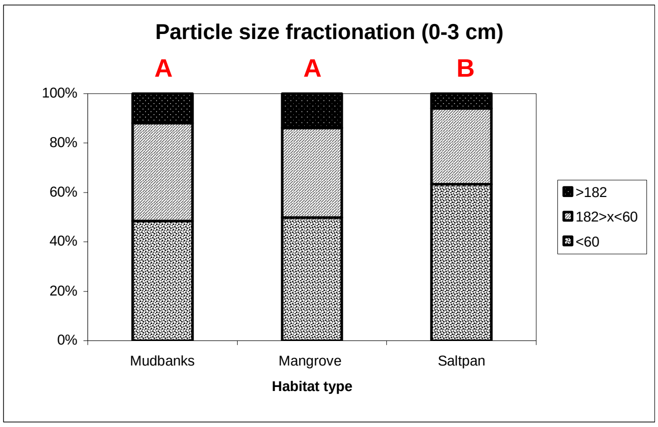


Figure 3.10. Transect particle size fractionation measurements <60 µm (surface 0–3 cm and mean depth profile) at Fisherman's Landing and Kangaroo Island

After particle size fractionation measurements from both locations were combined, saltflat habitat was found to contain higher proportions of sediment <60 µm than all other habitat types (Figure 3.11).



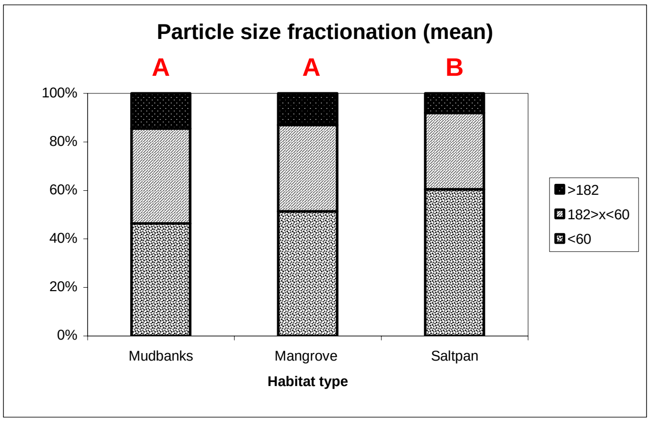


Figure 3.11. Mean surface and mean depth profile particle size fractionation measurements from each habitat type. Means with the same letter are not significantly different

One-way ANOVA analysis revealed that the mean proportion of sediment <60 µm was significantly different between habitat types for both the surface ($df = 37$, $F = 5.478$, $p = 0.009$) and mean depth ($df = 39$, $F = 9.284$, $p = 0.001$) measurements. Tukey's *post hoc* analysis found that the proportion of sediment <60 µm was significantly higher within saltflat habitat than both mangrove and mudbank habitats. Mudbank habitat in particular contained varying proportions of shell fragments, which contribute to the >182 µm size fraction.

As previously discussed, the particle size measurements do not correspond with porosity measurements as should be the case, in particular for saltflat habitat. Consequently, porosity was used as a surrogate for particle size comparisons with other analytes.

3.3.5 Total Kjeldahl nitrogen (TKN)

Apart from notable increases in TKN concentrations in near-shore mangrove habitat most likely due to enhanced sedimentation around fringes, TKN measurements remained relatively consistent along both transects for both depth measurements. The one obvious exception is the high surface saline grassland measurement of 2990 µg g^{-1} (dry weight) at the end of the Fisherman's Landing transect. This site also recorded the highest TOC and ammonium measurements and was sampled from within a vast cyanobacterial mat that clearly influenced the results (Figure 3.12).This trend was evident when TKN measurements from both locations were combined and compared between habitat types (Figure 3.13).

One-way ANOVA analysis revealed that mean TKN measurements were significantly different between habitat types for both the surface (*df* = 40, *F* = 15.021, *p* <0.001) and mean depth (*df* = 46, *F* = 16.610, *p* <0.001) measurements, following natural log transformation. Tukey's *post hoc* analysis found that TKN measurements were significantly higher within mangrove habitat than all other habitats. Seagrass TKN measurements were found to be significantly lower than all other habitats for both measurements.

In intertidal habitats the ratio of carbon to nitrogen is generally related. TKN is predominantly a measurement of organic nitrogen and therefore it is not surprising that mangrove habitat, which reported the highest TOC measurements, also reported the highest TKN concentrations. Conversely, the reverse applies for seagrass habitat.

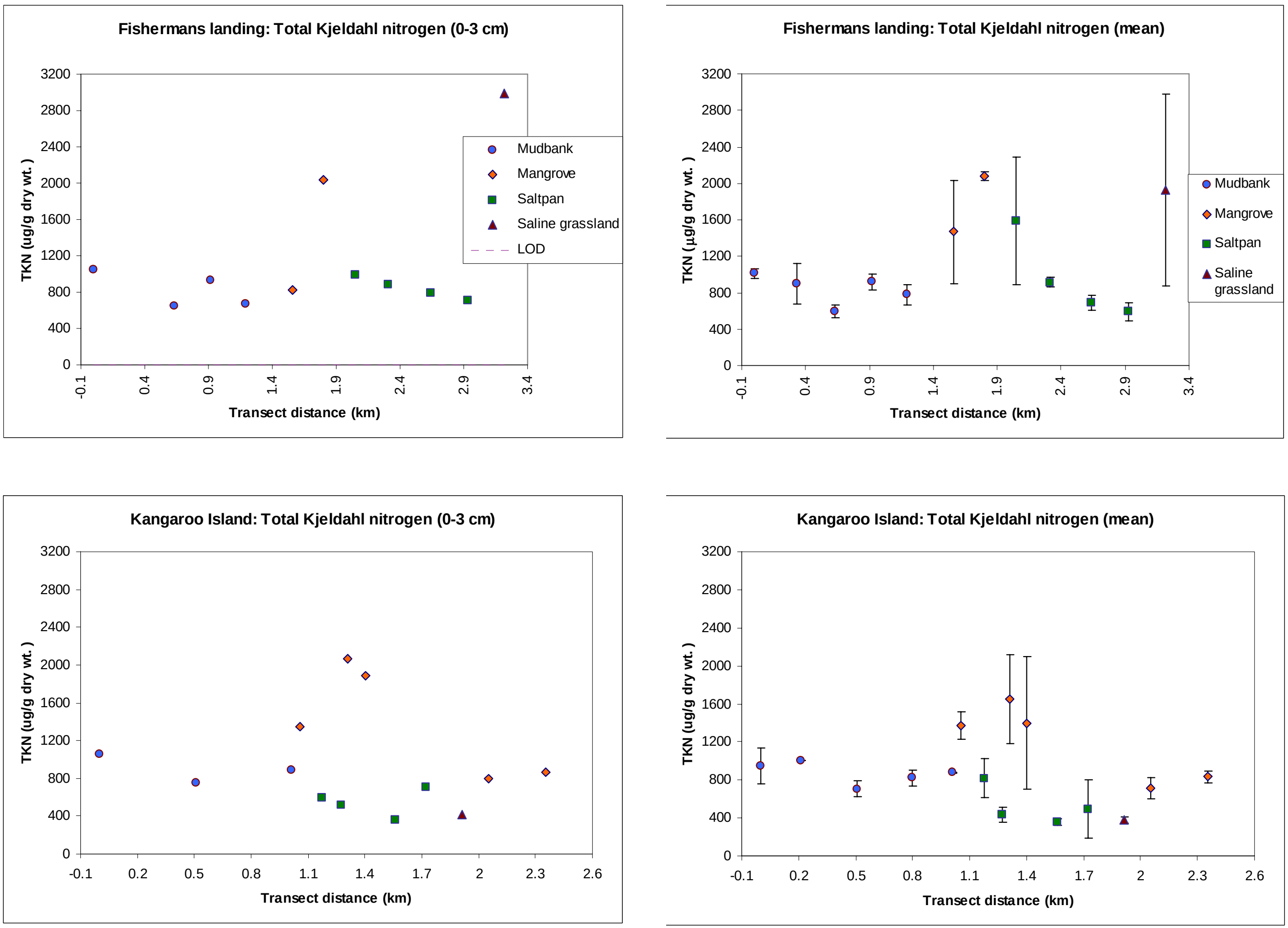


Figure 3.12. Transect total Kjeldahl nitrogen measurements (surface 0–3 cm and mean depth profile) at Fisherman's Landing and Kangaroo Island

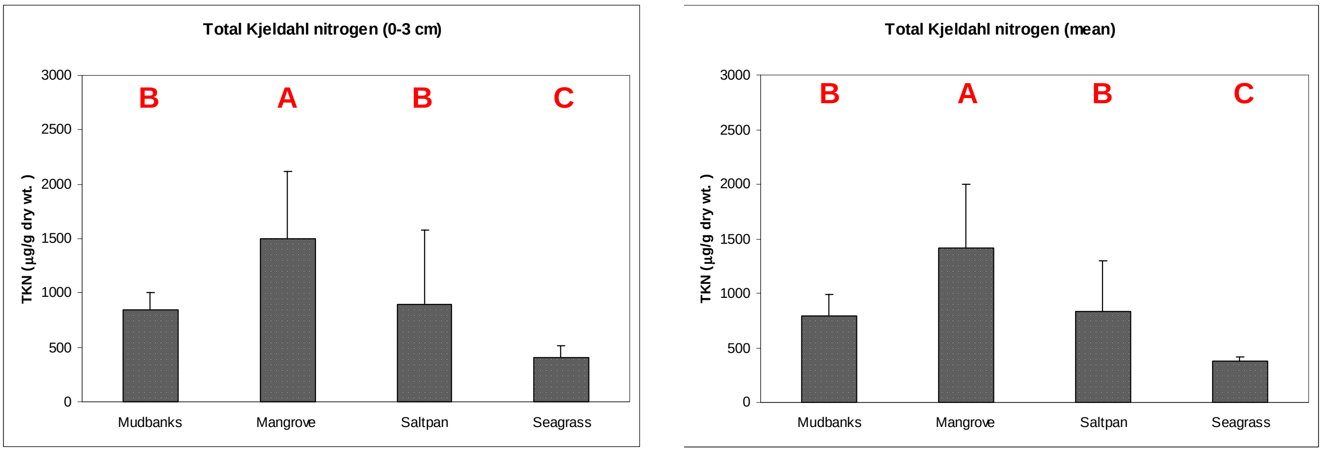


Figure 3.13. Mean (+1 s.d.) surface and mean depth profile total Kjeldahl nitrogen measurements from each habitat type. Means with the same letter are not significantly different

3.3.6 Ammonium (NH4+)

Ammonium measurements were found to generally decrease at Kangaroo Island after the second band of mangroves (i.e. ~1.6 km). No clear trends were apparent at Fisherman's Landing, however, although there was considerable variation within habitat types. Mean depth profile measurements at both sites indicate a relatively high degree of variation at different depths (Figure 3.14). Seagrass habitat reported the greatest differences in ammonium measurements between habitat types (Figure 3.15).

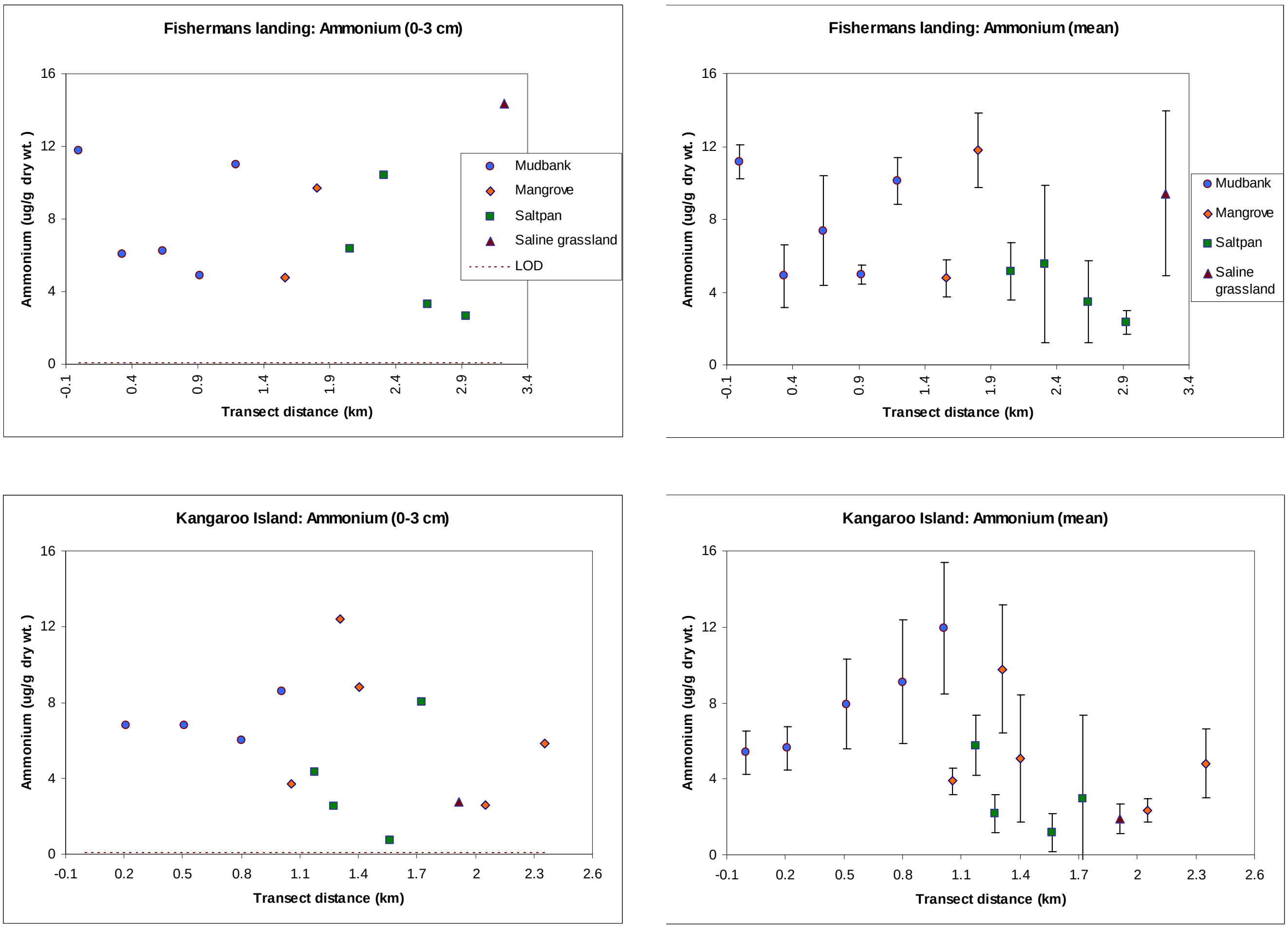


Figure 3.14. Transect ammonium (NH_4^+) measurements (surface 0–3 cm and mean depth profile) at Fisherman's Landing and Kangaroo Island

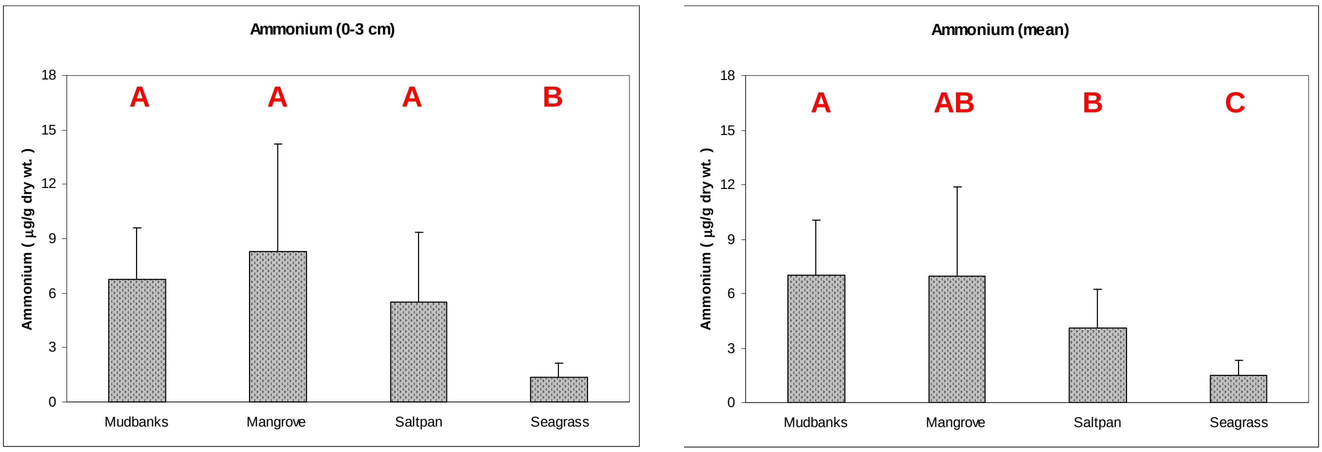


Figure 3.15. Mean (+1 s.d.) mean depth profile ammonium (NH_4^+) measurements from each habitat type. Means with the same letter are not significantly different

One-way ANOVA analysis revealed that mean ammonium measurements were significantly different between habitat types for both the surface (df = 43, F = 10.476, p <0.001) and mean depth (df = 47, F = 12.619, p <0.001) measurements, following natural log transformation. Tukey's *post hoc* analysis showed that surface ammonium measurements were significantly lower in seagrass habitat than all other habitats. Saltflat mean depth measurements were significantly lower than mudbanks and seagrass mean depth ammonium measurements were significantly lower than all other habitat types.

Typically, ammonium is derived primarily from anaerobic biological decay (Baird 1999) of readily biodegradable organic matter and therefore more anaerobic habitats and/or habitats with high organic content should report higher ammonium levels. Given the relatively low levels of organic carbon measured at seagrass sites, it is no surprise that ammonium concentrations were also correspondingly low. This is further good evidence that the seagrass beds have been devoid of grass for some time. It was, however, a little surprising that mean depth saltflat ammonium measurements were relatively low, as saltflat habitat was found to contain relatively high levels of TOC. Also, saltflats generally have fewer animal burrows compared with mangrove or mudbank habitats, thus decreasing the oxygen penetration. This may suggest that the TOC pool in mudbank habitat has a larger fraction of labile organic carbon than does saltflat habitat. It is also highly possible that the hypersaline conditions in saltflat habitat have considerably reduced oxidation of organic matter by the vast majority of Eubacteria species (Ollivier *et al*. 1994), thereby decreasing the amount of ammonium produced. It is not surprising that mangrove habitat reported relatively high concentrations of ammonium, particularly within the surface sediment, given the high concentrations of TKN and TOC.

3.3.7 Total phosphorous (TP)

TP measurements were relatively consistent between and within transect sites. The only exception can be seen at Kangaroo Island, where surface and mean depth TP measurements are observed to increase along the transect (~1.2 km) after the mudbank habitat, but decrease again after the second band of mangroves (>1.6 km) (Figure 3.16).

One-way ANOVA analysis found no significant differences in surface or mean depth TP measurements between habitat types.

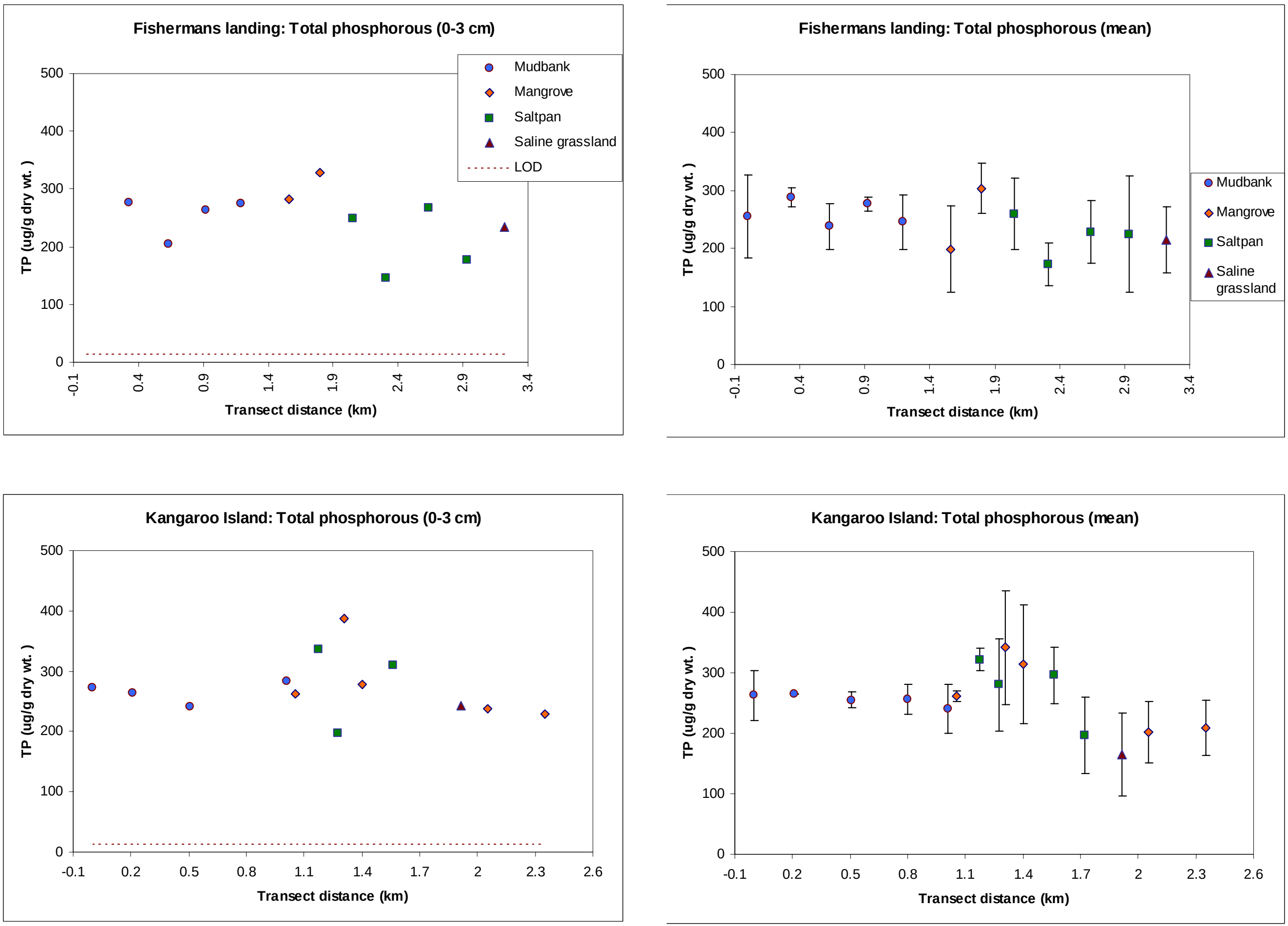


Figure 3.16. Transect total phosphorous (TP) measurements (surface 0–3 cm and mean depth profile) at Fisherman's Landing and Kangaroo Island

3.3.8 Exchangeable phosphate (Exchangeable-P)

Fisherman's Landing mean depth and surface exchangeable-P measurements were found to be generally higher within mudbank habitat than saltflat and relatively inland mangrove habitat. This trend was not as well defined at Kangaroo Island, where exchangeable-P measurements were found to be generally high within mudbank habitat but then also within saltflat and mangrove habitat (~1.3–1.6 km) (Figure 3.17).

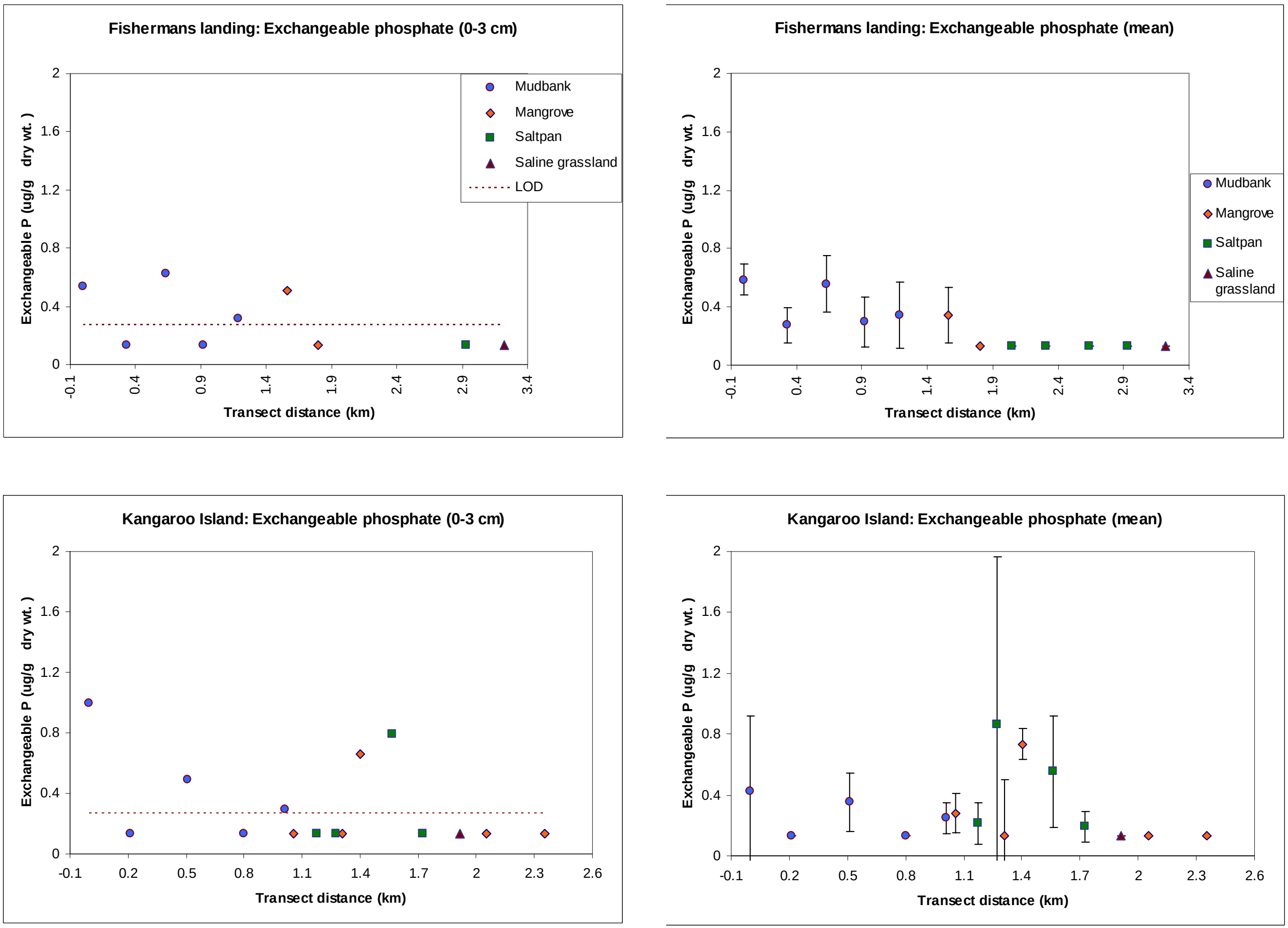


Figure 3.17. Transect exchangeable phosphate (exchangeable-P) measurements (surface 0–3 cm and mean depth profile) at Fisherman's Landing and Kangaroo Island

One-way ANOVA analysis found no significant differences in surface or mean depth exchangeable-P measurements between habitat types[7]. All measurements below the LOD (0.27 µg g^{-1} dry wt) were reported as half the LOD for statistical analysis. As ~50% of all measurements were found to be below the LOD, determining statistically significant differences was not possible.

The bioavailable concentrations of P (exchangeable-P) are very low relative to bioavailable N (ammonium) levels, which is interesting as this may suggest that P may be the limiting nutrient for primary production. It has usually been considered that N is the nutrient limiting plant growth in marine and occasionally in estuarine waters and sediments with low or variable salinity, whereas P is the limiting

[7] The deepest core depth measurement (6–9 cm) for the final saline grassland transect site at Fisherman's Landing (~ 3.3 km) was excluded from analysis. It is unknown whether this extremely high value of 21.82 µg g^{-1} (dry wt) is erroneous or not. It was not found to be a data entry error and is considered very unusual as the surface and 3–6 cm depth measurements from the same site were found to be below the detection limit. This was the same site that was covered by cyanobacterial mats and had elevated surface TN, TOC and ammonium measurements.

nutrient in freshwater ecosystems (Enell *et al*. 1995). However, the Peel Harvey Estuary in Western Australia was found to be potentially N-limited in summer and autumn but P-limited in winter and spring (McComb & Davis 1993). The ratio of N:P can therefore also be important and can determine what types of plant species proliferate.

The intracellular atomic ratio of N:P in algae and cyanobacteria is approximately 16:1. This ratio, also known as the Redfield ratio (Redfield 1958), is also close to that taken up by growing populations of these microphytes. Consequently, deviations from this ratio have often been used to evaluate the nutrient status of natural waters and sediments. When the proportion of N to P falls below the 16:1 level the ecosystem is said to be deficient in nitrogen. Conversely, values above 16 are phosphorous-deficient. Any ratio within 15–17:1 is considered to be not significantly different from the Redfield Ratio. Nitrogen-deficient conditions generally favour the growth of N-fixing cyanobacteria.

The sediment atomic ratio of NH_4^+:exchangeable-PO_4^{3-} [8] for mudbank habitat equals 105:1, mangrove (145:1), saltflat (87:1) and seagrass (22:1). Other studies (Atkinson & Smith 1984; Duarte 1990) have found this ratio to be higher than the Redfield ratio for vascular plants (24:1 and 20:1 respectively); however these results still suggest that phosphorous is the limiting nutrient in this ecosystem.

3.3.9 Acid volatile sulfides (AVS)

Mudbank habitat consistently reported the highest surface and mean depth AVS measurements along both transects. The only notable exception was the last mean depth mangrove site along the Kangaroo Island transect (~2.4 km, 24–27 cm deep), which measured 3.08 μmoles g^{-1} (dry wt) (Figure 3.18). Generally, although not exclusively, depth profile AVS measurements from the middle of the core were higher than surface measurements, which are thought to be relatively more oxic. Mid-core AVS measurements were generally higher than the deepest core measurements also, as deep core sulfides are gradually mineralised to form pyrite (FeS_2).

[8] These compounds of N and P are the most predominant forms bioavailable to organisms and therefore are used to calculate the molar ratios.

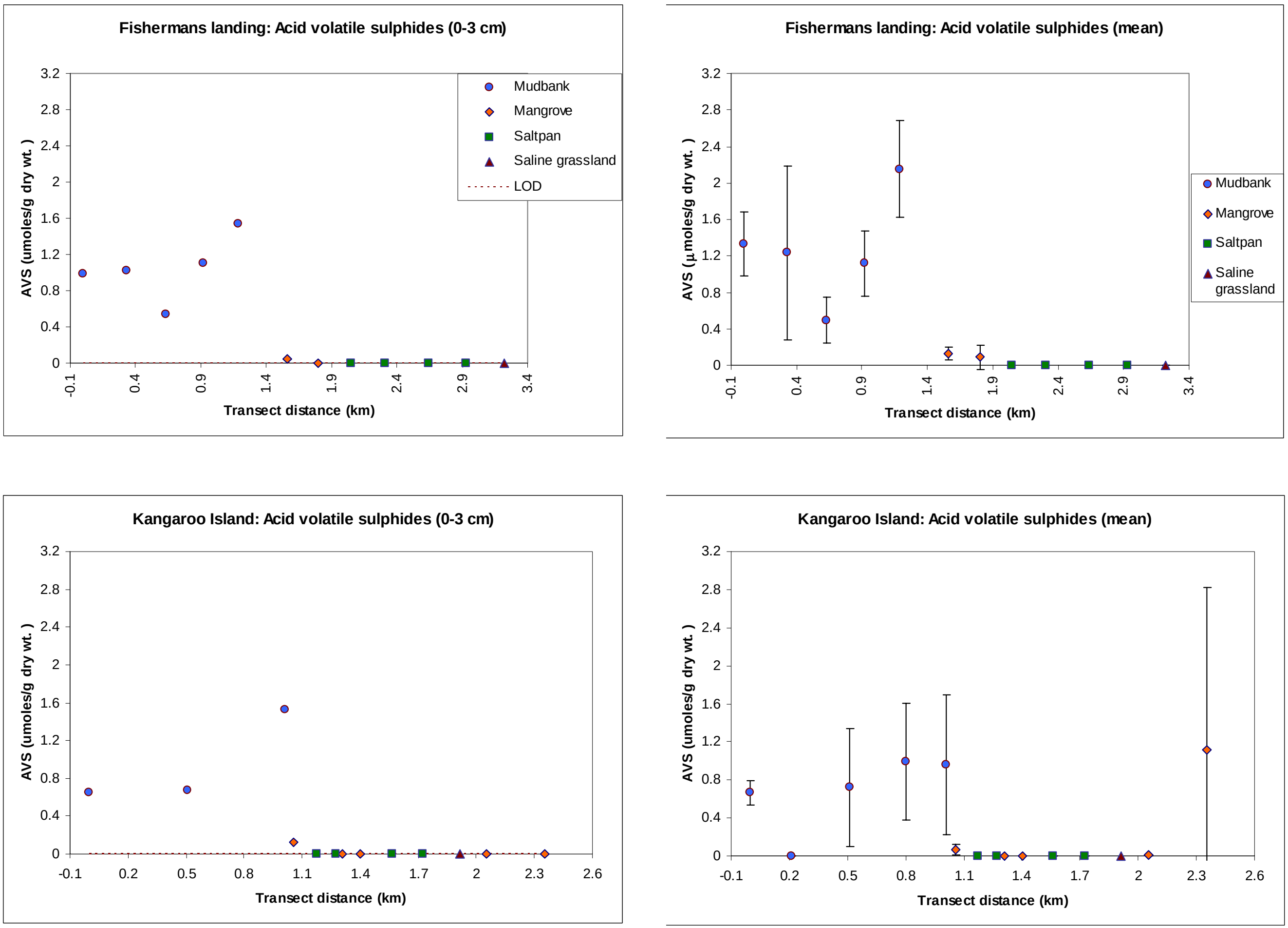


Figure 3.18. Transect acid volatile sulfides (AVS) measurements (surface 0–3 cm and mean depth profile) at Fisherman's Landing and Kangaroo Island

One-way ANOVA analysis revealed that mean AVS measurements were significantly different between habitat types for both the surface (df = 42, F = 3.070, p = 0.039)[9] and mean depth (df = 47, F = 9.521, p <0.001) measurements, following natural log transformation. Tukey's *post hoc* analysis showed that AVS measurements were significantly higher within mudbank habitat than saltflat habitat for both measurements. Additionally, mean depth seagrass measurements were significantly higher than saltflat. Many of the saltflat AVS measurements in particular were found to be below the LOD (0.002 µmoles g^{-1}) and therefore were expressed as half this value for statistical analysis.

[9] A more significant difference would have been apparent for the surface measurements if one of the random saltflat sites measuring 3.26 µmoles g^{-1} (dry wt) was excluded. This site was situated within a large patch of dead mangroves and it is thought that this unusually high AVS concentration is attributed to increased loads of labile organic material. Correspondingly high ammonia and TKN measurements support this notion.

This trend was again evident when AVS measurements from both locations were combined and compared between habitat types (Figure 3.19).

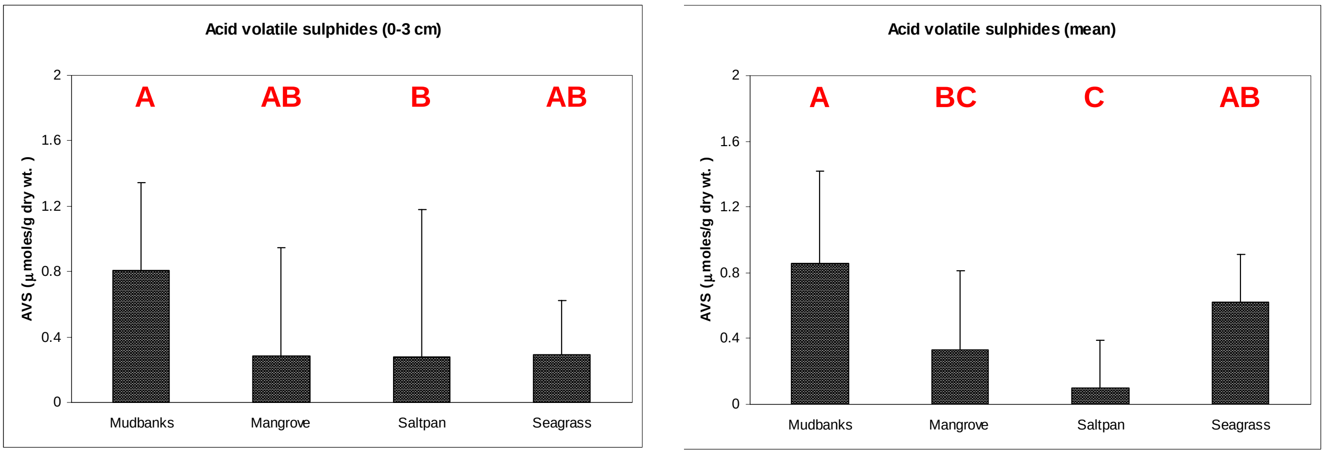


Figure 3.19. Mean (+1 s.d.) surface and mean depth profile acid volatile sulfides (AVS) measurements from each habitat type. Means with the same letter are not significantly different

These trends are mostly consistent with our expectations. It is thought that the hypersaline pore waters in the saltflat habitat may considerably inhibit metabolic processes of sulfate (SO_4^{2-}) reducing bacteria, to the point that virtually no sulfides accumulate in the sediment as has been found by Welsh *et al*. (1996). This was generally also found to be the case at the most saline, inland mangrove sites. Near-shore mangrove sites also reported lower AVS concentrations relative to mudbank sites, despite high TOC levels. It is thought that the net sulfide production may be close to zero because as quickly as SO_4^{2-} is reduced, sulfides may be oxidised by oxygen released by mangrove roots. Animal burrows and the dense penetration of mangrove roots into the sediment will also increase the aerobic layer considerably. Despite relatively lower TOC concentrations within mudbank habitat, it is thought that the oxic layer is smaller than that found in mangrove habitat (due mostly to the lack of roots) and therefore a net gain of SO_4^{2-} reduction at most sites was observed. At the time of sampling, seagrass habitat was thought to be relatively similar to mudbank habitat and therefore would be expected to have comparable AVS levels. It is plausible that the lower concentration of TOC recorded at seagrass sites would account for much of this difference in AVS concentrations.

3.3.10 Relationships between parameters

Significant relationships between the various parameters are displayed and discussed hereafter. The significance level was reduced to $\alpha = 0.01$ to avoid the possibility of type I errors. Significant relationships between analytes that do not interact were excluded (e.g. AVS and porewater salinity).

Proportional relationships were evident between total Kjeldahl nitrogen and total organic carbon for both measurements from all sites combined (Figure 3.20).

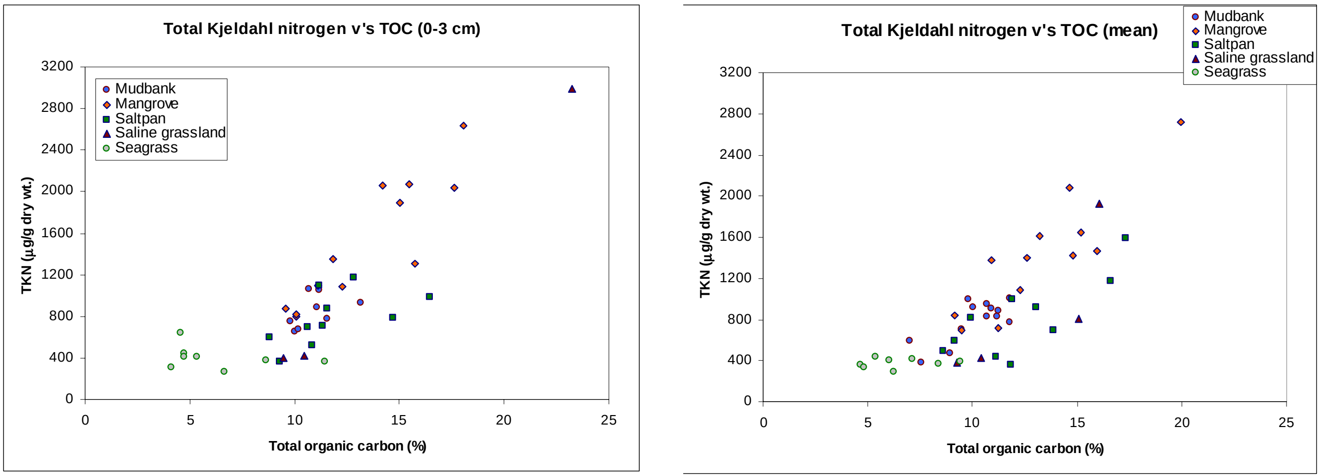


Figure 3.20. Relationships between total Kjeldahl nitrogen and total organic carbon for surface (0–3 cm) and mean depth measurements from all sites combined

Pearson's linear correlation analysis found TKN and TOC measurements to be significantly positively correlated from all sites combined, for both the surface ($n = 41$, $\alpha = 0.01$, $r = 0.769$, $p < 0.001$) and mean depth measurements ($n = 47$, $\alpha = 0.01$, $r = 0.801$, $p < 0.001$), following natural log transformation of both variables. Regression analysis indicated that 59% (surface measurements) and 64% (mean depth) of the variation in TKN is explained by variation in TOC ($r^2 = 0.591$ and 0.642 respectively).

As mentioned previously, a positive relationship between TKN and TOC was expected. The atomic ratio of TOC:TKN was found to be 150:1 for mudbank habitat, 109:1 for mangroves, 175:1 for saltflat and 204:1 for seagrass habitat, although these high C:N ratios are thought to be overestimated. The method used to determine TOC (fixed and volatile solids ignited at 550°C), also volatilises H, N and O and these losses in mass are included in the TOC measurement. The actual pool of C lost is typically about half that reported by the method. It should be noted that these molar C:N ratios are approximates only. These ratios do not reflect the bioavailable pool of C and N, as bioavailable C was not measured in this study. A large proportion of the TOC measurement is thought to contain recalcitrant C that is not readily available for biological degradation.

Mangrove communities are characterised by high primary productivity, mostly producing leaves. Upon decomposition, the intertidal leaf litter releases relatively high levels of organic carbon to the sediment. Given the high density of mangroves in the region, these high C:N ratios were not found to be unusual (Meziane & Tsuchiya 2000). It is interesting then that saltflat habitat reported the second highest C:N ratios, despite receiving tidal inundation only rarely (and therefore only occasional mangrove detritus input). This may be due to comparatively low levels of metabolic degradation of organic carbon in saltflat habitats, due to the hypersaline conditions. It is thought that mangrove detritus that has been transported offshore may contribute a major fraction of the organic carbon pool to mudbank and seagrass habitats. Residual cellulose and lignin from past seagrass beds would also add to this pool.

Proportional relationships were evident between ammonium and total organic carbon for both measurements from all sites combined (Figure 3.21).

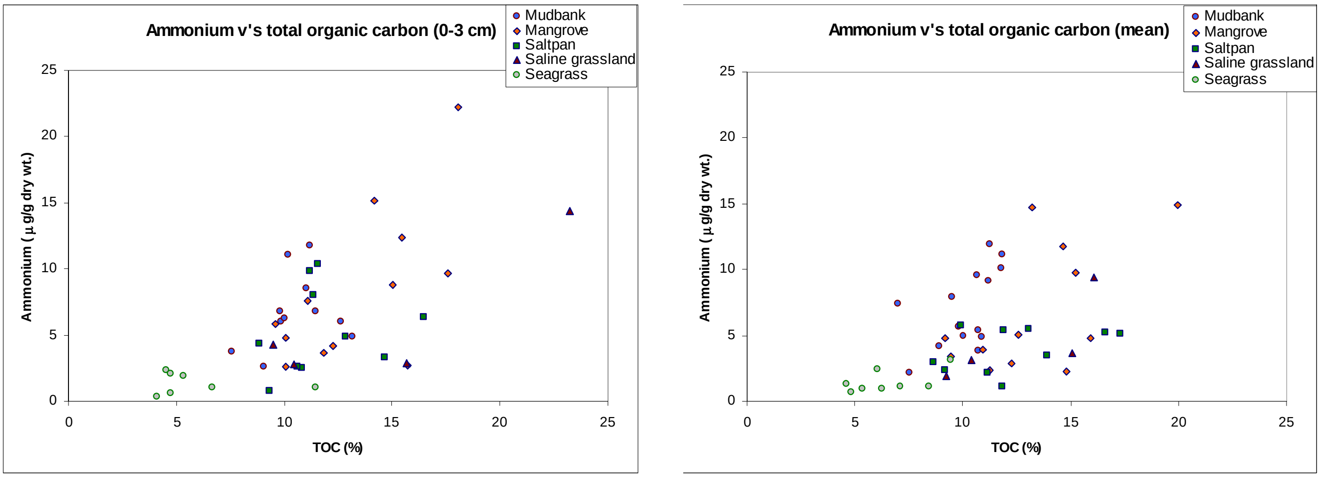


Figure 3.21. Relationships between ammonium and total organic carbon for surface (0–3 cm) and mean depth measurements from all sites combined

Pearson's linear correlation analysis found ammonium and TOC measurements to be significantly positively correlated from all sites combined, for surface (n = 44, α = 0.01, r = 0.659, p <0.001) and mean depth measurements (n = 48, α = 0.01, r = 0.617, p <0.001), following natural log transformation of both variables. Regression analysis indicated that over 43% (surface measurements) and 38% (mean depth) of the variation in ammonium is explained by variation in TOC (r^2 = 0.434 and 0.381 respectively).

This somewhat weaker relationship between ammonium and TOC (rather than TKN and TOC) was expected, as sediment ammonium is a degradation product of the labile organic carbon fraction, which is only a proportion of the TOC measurement.

A proportional relationship was also evident between total Kjeldahl nitrogen and porosity for both measurements from all sites combined (Figure 3.22).

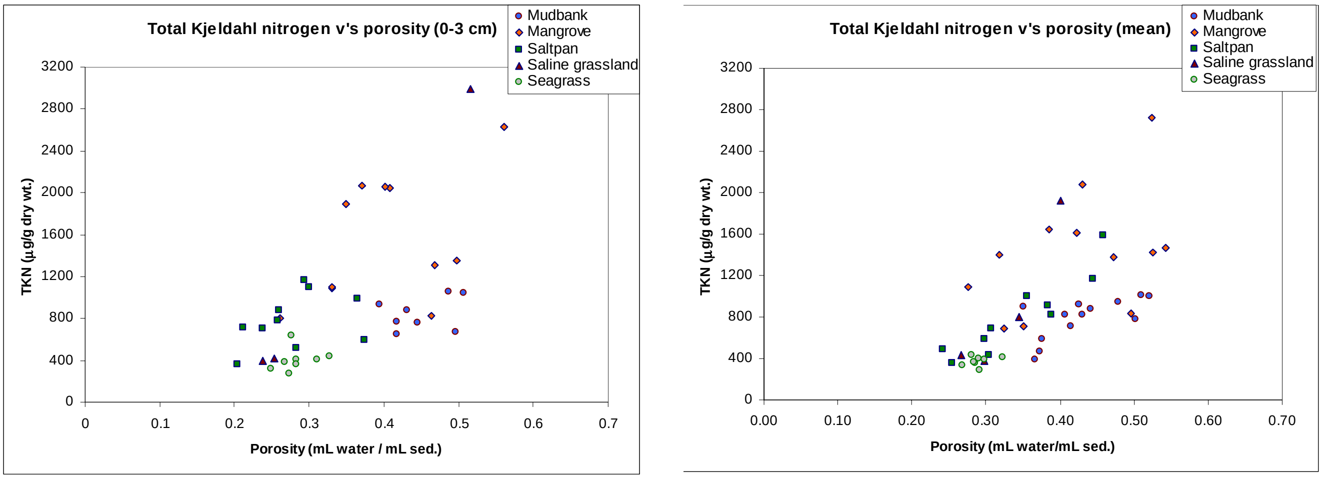


Figure 3.22. Relationships between total Kjeldahl nitrogen and porosity for surface (0–3 cm) and mean depth measurements from all sites combined

Pearson's linear correlation analysis found TKN and porosity measurements to be significantly positively correlated from all sites combined, for both the surface (n = 40, α = 0.01, r = 0.625, p <0.001) and mean depth measurements (n = 47, α = 0.01, r = 0.705, p <0.001), following natural log transformation of TKN. Regression analysis indicated that 39% (surface measurements) and almost 50% (mean depth) of the variation in TKN is explained by variation in porosity (r^2 = 0.391 and 0.497 respectively).

Using porosity as a surrogate for particle size fractionation, these relationships are expected, as finer sediments have a larger exchange capacity for all cations and anions. Not surprisingly, this relationship was also significantly positively correlated between ammonium and porosity as NH_4^+ is the dominant ionic form of N which would bind to surface exchange sites on sediment particles (surface: n = 43, α = 0.01, r = 0.596, p <0.001; mean depth: n = 48, α = 0.01, r = 0.667, p <0.001), following natural log transformation of ammonium. Similarly, this relationship was also significantly positively correlated between TOC and porosity for both the surface (n = 47, α = 0.01, r = 0.402, p = 0.005) and mean depth measurements (n = 48, α = 0.01, r = 0.520, p <0.001), following natural log transformation of TOC.

Proportional relationships were evident between total Kjeldahl nitrogen and ammonium for both measurements from all sites combined (Figure 3.23).

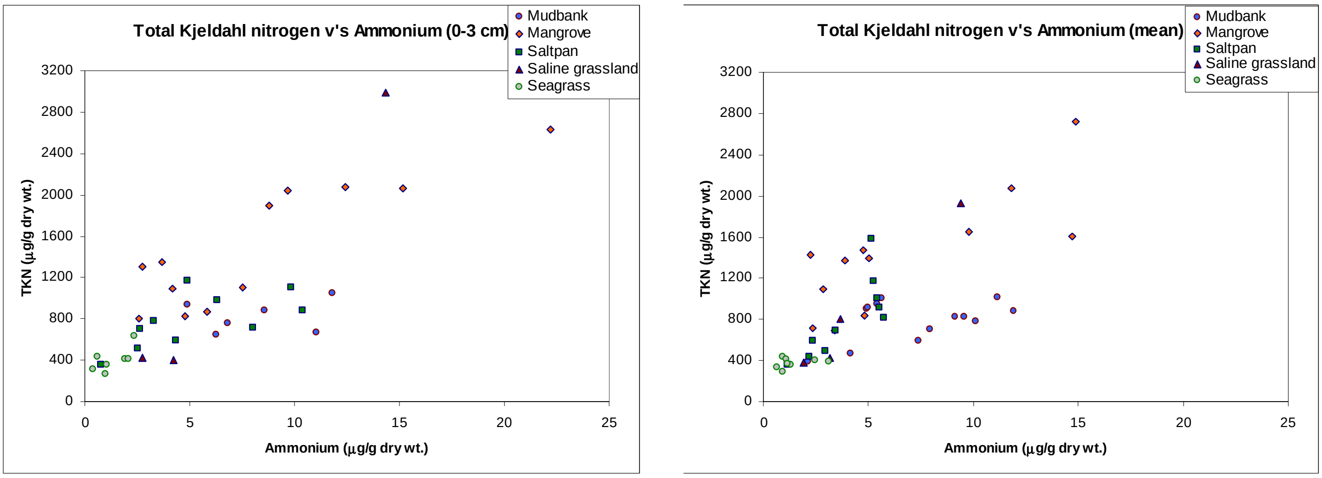


Figure 3.23. Relationships between ammonium and total Kjeldahl nitrogen for surface (0–3 cm) and mean depth measurements from all sites combined

Pearson's linear correlation analysis found ammonium and TKN measurements to be significantly positively correlated from all sites combined, for both the surface (n = 38, α = 0.01, r = 0.806, p <0.001) and mean depth measurements (n = 47, α = 0.01, r = 0.755, p <0.001), following natural log transformation of ammonium and TKN. Regression analysis indicated that 65% (surface measurements) and 57% (mean depth) of the variation in ammonium is explained by variation in TKN (r^2 = 0.650 and 0.570 respectively). These relatively robust relationships are of no surprise; TKN and ammonium measurements would normally be proportionally related, as ammonium is generated during microbial decomposition of the organic fraction of the TKN pool.

3.4 Conclusions

In general, the relationships between the given sediment parameters were to be expected. Porosity and TOC were found to be closely related to TKN and ammonium measurements. The ratios of C:N:P appear to be quite high, suggesting that the sediment may be phosphorous-deficient. The concentrations of organic carbon are not unusual, given the high density of mangroves in the region.

Hydrological and sedimentation dynamics are the most important factors governing subtidal (or low intertidal) habitat zonation. Porewater salinity was not found to differ from the water column salinity for either mudbank or seagrass habitat and consequently did not relate to habitat zonation. Although hydrological and sedimentation parameters were not investigated in this study, other studies have found that shelter, slope and the rate of sediment supply are the principle factors that determine the distribution of subtidal habitats. Therefore, changes in the coastal hydrodynamics would impact on sediment deposition and potentially affect seagrasses, in particular. Within the subtidal zone, seagrasses may colonise areas that are favourable for photosynthesis, settlement and recruitment and are deep enough to minimise desiccation stress. The seagrass sites investigated in this study were found to be atypical of seagrass habitat. Generally, seagrass habitat will have high TOC and nutrient levels and the presence of seagrass will lead to settlement of very fine sediment (therefore high porosity), thus promoting further colonisation. As previously discussed, the seagrass sites sampled in this study were found to be devoid of seagrass and therefore without stabilisation by the roots, the overlying fine sediment has eroded away. This has exposed the coarser underlying sediment that was found to be deficient in organic and inorganic nutrients and organic carbon. Seagrasses are unlikely to become re-established where this has occurred.

The literature has reported that physicochemical parameters are important in determining intertidal vegetation community zonation and these results confirm this notion. Porewater salinity in particular was found to be the most important parameter studied. Results indicate that salinity directly determined the dominant vegetation type of the intertidal zone and at high salinities dramatically reduced microbial metabolic activity, consequently affecting concentrations of other parameters such as AVS, ammonium and TOC. The salinity gradient became increasingly evident moving inland along both transects, but Kangaroo Island in particular. This gradient is directly affected by the frequency of tidal inundation and subsequent evaporation and the frequency and volume of terrestrial freshwater runoff.

It must be remembered also that simply the presence (or absence) of vegetation for a given habitat type will drastically affect the concentration and relationships of a number of sediment parameters. For example, mangrove roots will oxygenate the sediment, which will decrease the sulfide concentration, which in turn will make conditions more favourable for the growth of mangroves. As previously mentioned, the absence of seagrass in this study greatly affected the seagrass sediment measurements.

The main implication for environmental managers within Port Curtis is that hydrological connectivity is critical to the protection of habitat. The health and ultimately the distribution and extent of different vegetation types depend on the extent of tidal inundation. The processes determining where intertidal ecosystems develop appear to be the same as in other tropical regions. Future studies on the water quality (nutrients and suspended sediments in particular) and seagrass dynamics would be useful to pick up changes within the Port Curtis intertidal ecosystem.

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Chapter 4. Historical trends in recreational fishing catches in the Gladstone region

John R. Platten

4.1 Introduction

Trends in the catches of amateur fishers are regarded as indicators of both ecologically sustainable development (see EPA, NSW 1995) and the degree of exploitation of fish (Gartside *et al*. 1999). Recreational fishing is one of the highest participation activities in Queensland and its contribution to the Queensland economy is significant. As a result of this participation rate, the catches of anglers can influence government policy: when anglers are increasingly dissatisfied, they have a large political influence for change. Therefore, quantitative knowledge of catch trends is very important in informing debate.

Recreational catch data can also be useful as a monitoring tool. Data from a group of fishers going to the same place, on comparable tides, and fishing in the same way for a set time period can be a very cost-effective manner of examining ecological productivity trends over time and may be almost as useful as some scientifically designed sampling programs performed at large cost.

In some cases fishers have been fishing in comparable ways at particular sites over significant time periods and have maintained careful records of their catches (see Pollock & Williams 1983). This provides useful trend data provided that the characteristics of the fishery are well understood (Gartside *et al*. 1999). In many cases, recreational catch data are the only reliable long-term data available that may show change over time.

This study examined trends over time in two data sets related to the records of line-fishing clubs. One was based on an inshore fishery in the estuaries near and in Port Curtis, the other in offshore waters. Both sets provide a point of comparison to other data and studies elsewhere in Queensland.

It is not the aim of this paper to definitively establish the reasons for any of the trends in catches. This is a separate and highly complex task that requires much further study. Caution must be applied in postulating the causes of trends, as the factors that impact on recreational fisher success are many and varied. It must be observed, however, that many of these factors apply equally to other forms of monitoring so that there is merit in examining trends at least to identify if there is cause for concern.

Care should also be taken in comparing the data from the two clubs. The characteristics of the offshore fishery are fundamentally different from those of the inshore fishery. The data from the Yaralla club are based on large predatory fish in a reef environment, the Wanderers club on inshore estuarine species. The two sets are provided to demonstrate separate trends in different habitats, not for direct comparison.

4.2 Data sets examined

4.2.1 Wanderers fishing club

These fishers conduct monthly competitions in estuaries from Bustard Head to Cape Capricorn and hold reliable data from 1982–83, 1987–88 and from 1990–91 to the present. Each club trip is held over five hours in conjunction with a spring tide. They fish in similar locations each year. Hence there are data available in each year from catches taken from the shores of Facing Island within Gladstone Harbour, and from the beaches close to Cape Capricorn (Figure 4.1).

Fishing has always been conducted using rod and reel techniques, and usually natural baits of yabbies (*Callianassa* spp.) are utilised. All competitors fish the same locality and for the same length of time.

The data contains a degree of control in that the fishers fished:

- in set comparable locations,
- over set time periods,
- on days with similar tides,
- using similar methods, and
- seeking similar species.

Some factors are not controlled. The skill level of competitors will vary and the group fishing from year to year does not remain the same. However, a core group of fishers have remained active throughout much of the study period. There are controls on the type of gear that can be used but increasingly efficient gear (or experience in using it) could influence results.

The club records the number of fish of each species taken and the total weight of the catch for each angler.

4.2.2 Yaralla fishing club

The Gladstone-based Yaralla fishing club provided data for the period 1977 to 2000 based on catches from the Capricorn Bunker group (Figure 4.1). Each trip consists of a 2-day weekend excursion with anglers fishing for approximately 10 hours per trip. Fishing is carried out in daylight from vessels from 10 m to 17 m in length. In almost all cases, fishing occurred from boats drifting across patch reefs in depths between 12 m and 25 m. Fishing is carried out with nylon handlines (rods and reels are banned from competition) using baits of fish, squid or cuttlefish.

The data are from experienced anglers fishing in known successful locations, so their catch rates are not indicative of those of most anglers. They are, however, useful to indicate trends in catch rates over time. The fishers are competitive, hence there is a motivation to be as productive as possible and to record catches accurately.

The Yaralla club has always recorded the total number and weight of fish caught for each angler but the catch has only been recorded by species since 1997.

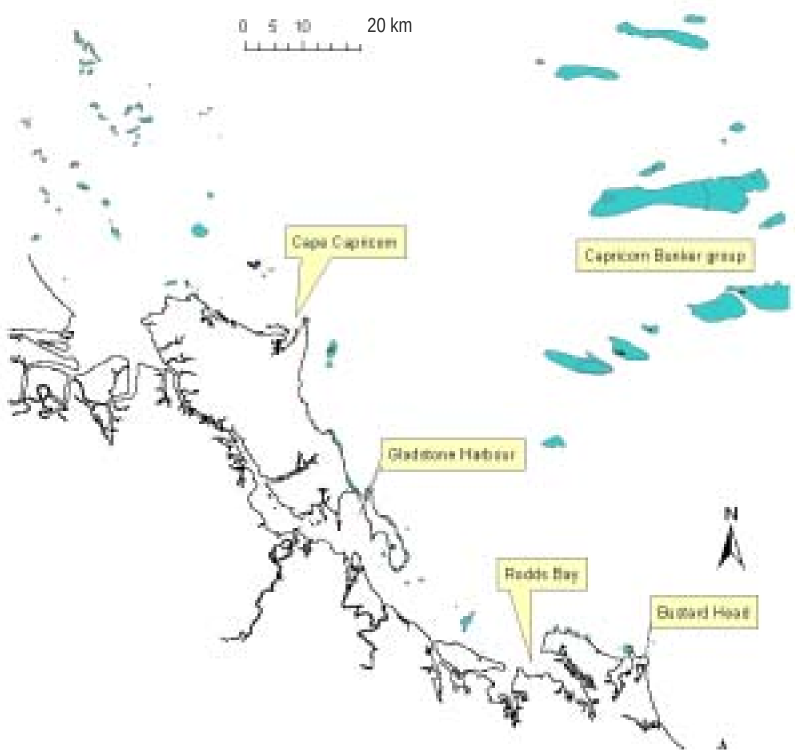

Figure 4.1. Locations fished by the Wanderers and Yaralla fishing clubs

4.3 Methods

Data were analysed to indicate the median number of fish caught per angler per trip ($fish.angler^{-1}.trip^{-1}$) for each financial year (July to June) at each locality. The median was chosen to reduce any bias of unusually low or high individual catches and variations in skill between anglers (see Mapstone *et al*. 1996). Trends in species composition of catch were examined by totalling the number of each species taken in each year.

Trends in catch rate over time were estimated by examining the correlation between median catch and the time in years since the first year data were available, thereby attempting to examine the degree to which time can explain trends in median catch.

4.4 Results

4.4.1 Catch composition

The catch of Wanderers fishing club is dominated (81% in Gladstone Harbour, 85% at Cape Capricorn) by whiting (largely *Sillago ciliata*) (Figure 4.2). The catch from the Yaralla club is based around three main species [red throat emperor (*Lethrinus miniatus*.), venus tusk fish (*Choerodon venustus*) and coral trout (*Plectropomus* spp.)], although a large number of other species are taken (Figure 4.3).

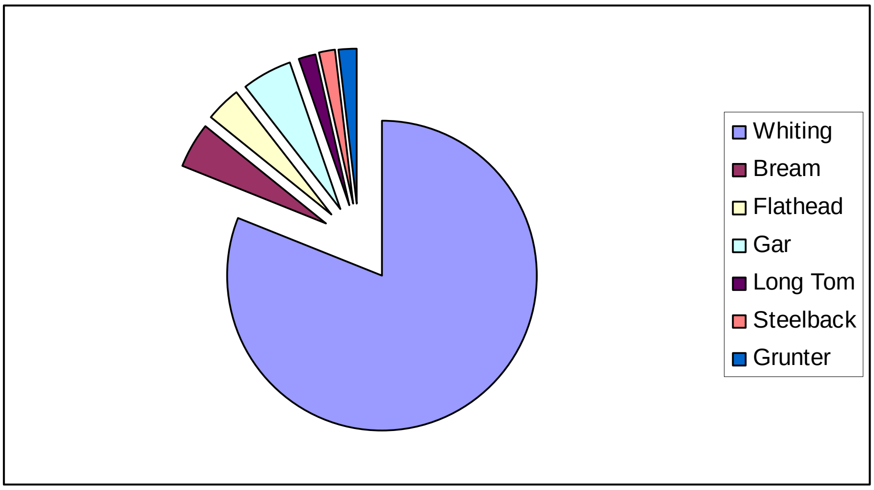

Figure 4.2. Catch composition of the Wanderers fishing club from the Curtis Coast area, 1997–2000

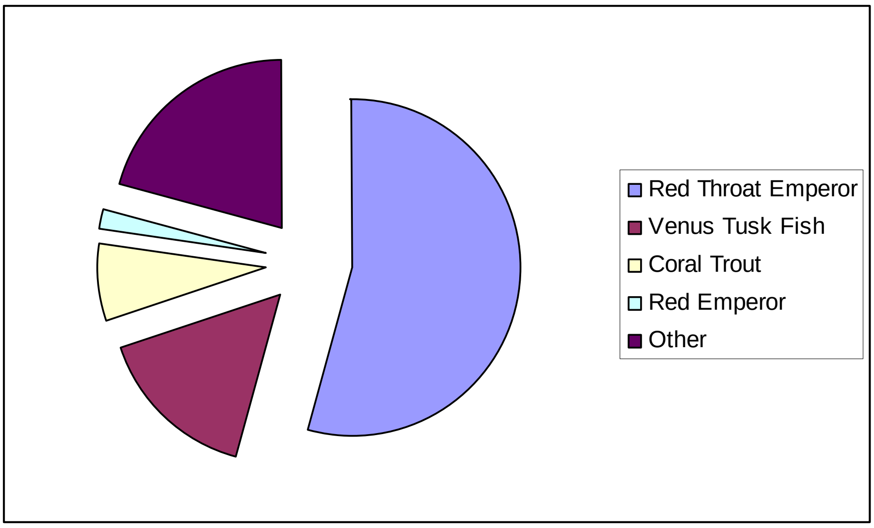

Figure 4.3. Catch composition of the Yaralla fishing club from the Capricorn Bunker group, 1997–2000

4.4.2 Catch trends

Wanderers fishing club

Highest catch rates were consistently recorded from the Cape Capricorn area (Figure 4.4, Table 4.1). Trends over time (whole of time period and last ten years) were examined by calculating the correlation between time and catch rate. Catch rates were variable, but show a significant declining trend across the full time examined at both the Cape Capricorn and Gladstone Harbour sites (Table 4.1).

During the period 1991–2001, this trend was less obvious at Cape Capricorn, but catch rate continued to decline within Port Curtis (Table 4.1).

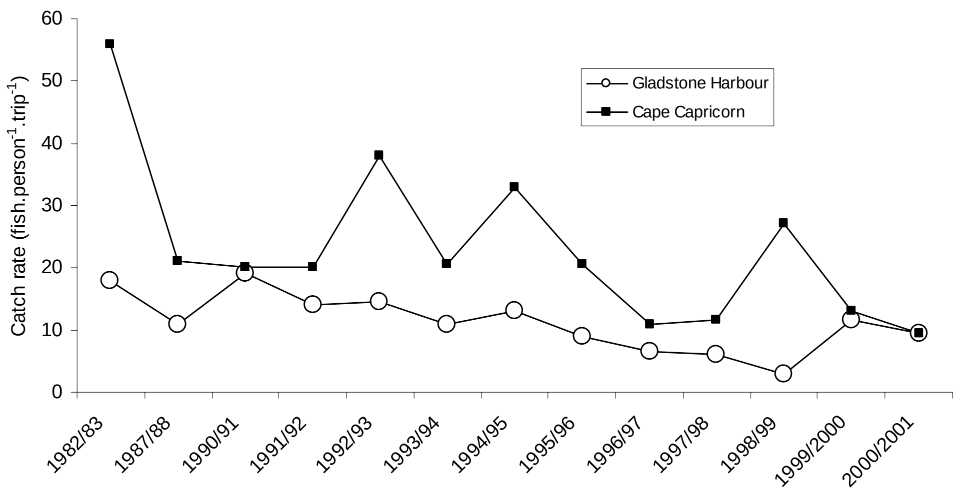

Figure 4.4. Catch rates by the Wanderers fishing Club (median catch per person per trip) from Gladstone Harbour and Cape Capricorn, 1982–2001

Table 4.1. Catch rates (median catch per person per trip) of the Wanderers fishing club for the two principal locations (Cape Capricorn and Gladstone Harbour), for the period 1982–1983 to 2000–2001

	Gladstone Harbour	Cape Capricorn
1982–83	18	56
1987–88	11	21
1990–91	19	20
1991–92	14	20
1992–93	14.5	38
1993–94	11	20.5
1994–95	13	33
1995–96	9	20.5
1996–97	6.5	11
1997–98	6	11.5
1998–99	3	27
1999–2000	11.5	13
2000–2001	9.5	11.5
Trend over entire period	r = -0.70	r = -0.71
Trend 1991–2001	r = -0.73	r = -0.32

Yaralla fishing club

No significant trend was detected for offshore catches from the Yaralla fishing club (Figure 4.5; r^2=0.12, p>0.05). Variability from year to year is apparently more important than trends over time and the probable best interpretation is that catch rates have not significantly declined over time.

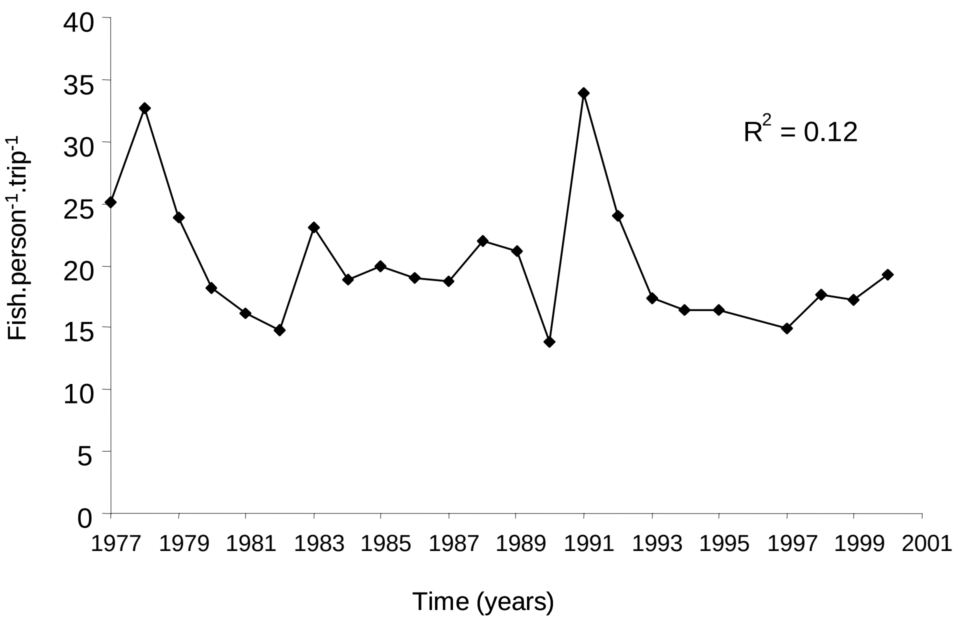

Figure 4.5. Catch rates (median catch per person per trip) of the Yaralla fishing club for offshore locations within the Capricorn Bunker group, 1977–2000

4.5 Discussion

The data provide a useful opportunity to examine trends in fish catches at a number of sites within the vicinity of the Curtis coast. They represent one of the few long-term data sets available to consider possible trends in ecological productivity.

One other dataset exists that measures time-series ecological productivity within Gladstone Harbour. Currie and Small (2005) examined trends in macrobenthos abundance and species richness at a number of sites within the harbour over the period 1995 to 2000. This study is of significance to the current paper because the diet of whiting consists largely of benthic invertebrates (Burchmore *et al*. 1988), so that an ecological link could exist between them. To investigate any link between the two datasets, trends in macrobenthos abundance (Table 1 of Currie & Small 2005) and catch rates of the Wanderers fishing club (described above) were compared to determine any correlation. A Pearson correlation matrix comparing the measures shows that they were significantly related (r^2=0.97, $p<0.001$) as seen on a time-series plot of the two measures (Figure 4.6).

There are two major implications from this relationship. The first is that catch rates may reflect broader issues related to ecological productivity and may not be simply reflecting fishery-dependent issues such as overfishing. This deserves closer attention. The second is that the decline in catch rate could reflect broader decline in ecological productivity. This too, is also a cause for concern and warrants further research effort.

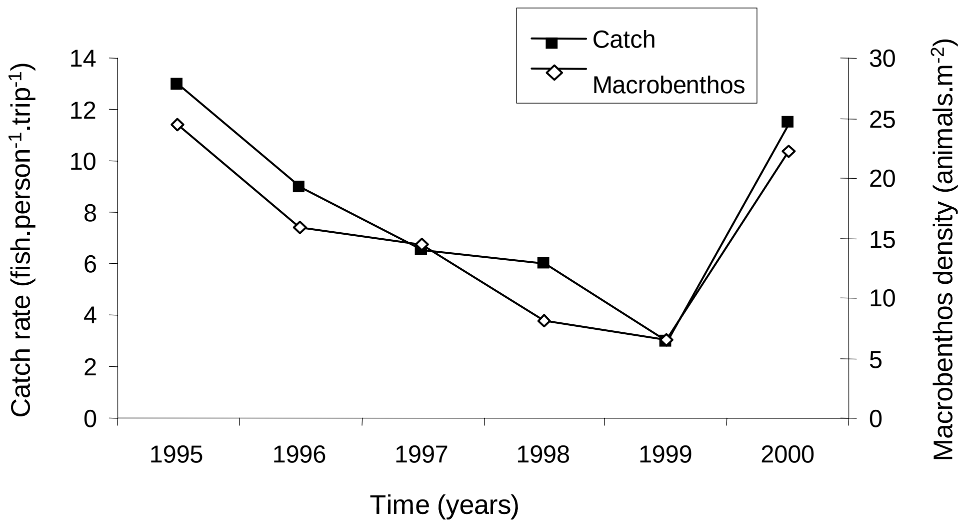

Figure 4.6. Relationship between the catch rates of the Wanderers fishing club and macrobenthos abundance, 1995–2000. Macrobenthos abundance data are after Table 1 of Currie and Small (2005)

As stated in the introduction to this chapter, this report does not establish any causative relationship behind catch trends and no inferences in this regard should be drawn directly from this study. However, it is notable that habitat modification has been historically much greater within Gladstone Harbour than at Cape Capricorn or within the Capricorn Bunker group. This, too, is a subject worthy of further investigation.

Another possible cause for catch decline in Gladstone Harbour is considered in the CRC flows research report (Halliday *et al*. 2005). There the correlation between river flows and catch rates described here was examined in detail. Figure 4.7 shows the relationship between catch rates and combined river flows for the Calliope and Boyne Rivers. It is hypothesised that total flows greater than 150 000 ML positively influence catch rates, since a significant positive correlation is evident for flows larger than this ($r=0.99$, $p<0.001$) but not for smaller flows ($r=-0.13$, $p>0.05$).

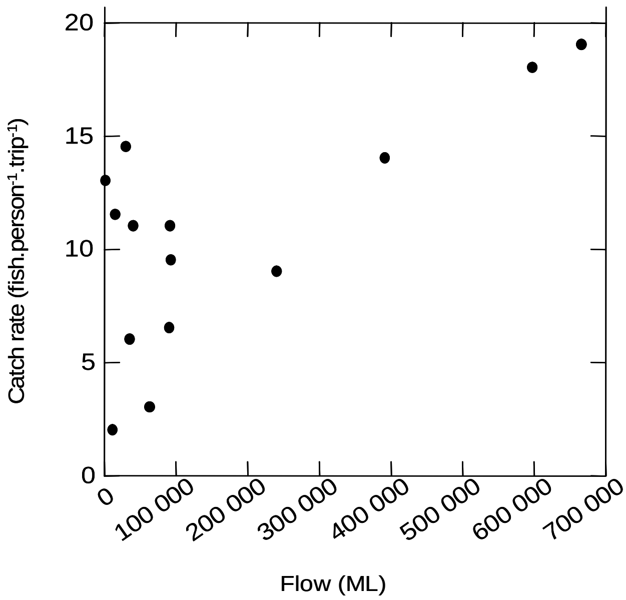

Figure 4.7. Relationship between the combined flow of the Boyne and Calliope Rivers and catch rates of the Wanderers fishing club from Gladstone Harbour, 1983–2002

In the flow study, it was demonstrated that a similar threshold flow relationship may exist between the catch at Cape Capricorn and flows from the Fitzroy River. Cape Capricorn catches appear influenced by Fitzroy flows greater than 1 GL (1 000 000 ML). The Pearson correlation between flows and catch for flows above 1 GL was significant ($r=0.91$, $p<0.05$). It is noteworthy that only two combined flow events greater than 150 000 ML have occurred in Gladstone Harbour in the 10 years after 1991, but seven of the ten years after 1991 have seen Fitzroy flows greater than 1 GL. This could (at least in part) explain why the fall in catch rate in Gladstone Harbour has been greater than at Cape Capricorn.

This relationship is of considerable importance, particularly since flow regulation of the Boyne River may decrease the frequency of flow events, and the projections of the impact of climate change on river flows also points to decreasing river flows. If catch rates are indicative of estuarine productivity, then there is obvious cause for concern. The link between estuarine and inshore productivity and climate change has received little research attention and deserves much closer examination.

Care must also be expressed in considering catch trends over time. Some trends established in this study were clearly significant but there is no guarantee that catch rate trends may not change over a longer time period. This is why it is important to consider catch rates over as long a time period as possible.

Declining trends were common to all inshore sites over the time that data was available, although the trend was much less significant in the catch rates from Cape Capricorn over the last ten years. There was, however, evidence of a continuing decline within Gladstone Harbour. This trend warrants further investigation and perhaps fishery-independent monitoring.

Of further interest is that there is no clear trend in catches from the reef areas of the Capricorn Bunker group; variability from year to year was more significant than any trend over time. This suggests that declining catch rates are not universal among all local recreational fisheries.

The study of Thwaites and Williams (1994) of southern Queensland fishing club data provides a comparison in relation to the results of the Wanderers club. The 1994 study examined the fishing club results from several clubs using similar techniques, catching similar species and operating under similar competition rules within Moreton Bay in southern Queensland. Their study showed no declining trend at four of five sites (all have remained stable or increased slightly). One site (Bribie Island) showed a decline in catch rates; however the significance of the decline was lower than that recorded in this study from Gladstone Harbour. This suggests that the trends observed in this study are not necessarily evident in other locations of the state.

Higgs (1993) studied trends in the data of fishing clubs fishing Great Barrier Reef waters offshore from Townsville. These anglers fished in a similar manner to the Yaralla club and their catch was based around similar species. Higgs records a complex but declining trend in catch. The trend in catch rate from the Capricorn Bunkers was nowhere near as obvious. A slight decline was detected but variation from year to year was such that catch could not be considered to have declined significantly.

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Chapter 5. Distribution and assemblage composition of demersal fish in shallow, nearshore waters of Port Curtis

David R. Currie and Rod M. Connolly

5.1 Introduction

Estuaries are widely recognised as important nursery grounds for juvenile fish, and are thought to provide a high availability of food resources and offer shelter from predators (Blaber 1980; Robertson & Duke 1987; Blaber *et al*. 1989). Human alterations to estuaries, including loss of habitat, may therefore have profound effects on fishery productivity and other ecological processes (Nichols *et al*. 1986; Hutchings 1999). Land reclamation of intertidal areas along the Forth estuary in Scotland have, for example, resulted in a loss of over 50% of the invertebrate benthic production, and reportedly led to large declines in fish and bird populations (McLusky *et al*. 1992). Characterising and evaluating the relative importance of benthic habitats is therefore a fundamental requirement for the conservation and better management of these waterways.

This study concentrates on assessing species composition and distribution of demersal fish in a marine-dominated estuary, and their utilisation of different sedimentary facies. In particular, we examine 1) spatial patterns in the community structure of fish and their relationships with prey availability, 2) the importance of structural complexity (i.e. seagrass availability) in influencing species composition, and 3) the relative importance of a range of environmental variables (including sediment structure, depth, salinity and seagrass biomass) in governing the abundance and diversity of fish.

5.2 Methods

5.2.1 Trawl sampling

Fish were collected from 105 intertidal and shallow subtidal sampling stations [<5 m below mean low water springs (MLWS)] using a 5 m beam trawl net with 2.5 cm mesh (Figure 5.1). These stations were spatially matched with locations sampled previously for macrobenthos and water quality, and collectively included a random selection of riverine, estuarine and open coastal habitats. Three replicate 200 m trawl shots were undertaken at each station, and individual trawl tracks recorded using a differential GPS. The speed of the net over the ground (1.8–2.5 km) was later estimated from the latitude and longitude at the start and finish of each shot and the duration of each shot. All sampling was conducted during daylight hours between 17 November and 5 December 2003. Where possible all trawl shots were directed into the prevailing current. The entire catch from each shot was stored on ice and subsequently frozen prior to laboratory analysis. Three green turtles *Chelonia mydas* inadvertently captured during the survey were returned to the water alive. All fish, invertebrates and sea snakes collected were identified to species level before being counted, measured [total length (mm) fish only] and weighed (g). In addition, representative tissue samples and gut contents from all species were retained and archived for prospective studies of estuarine trophodynamics.

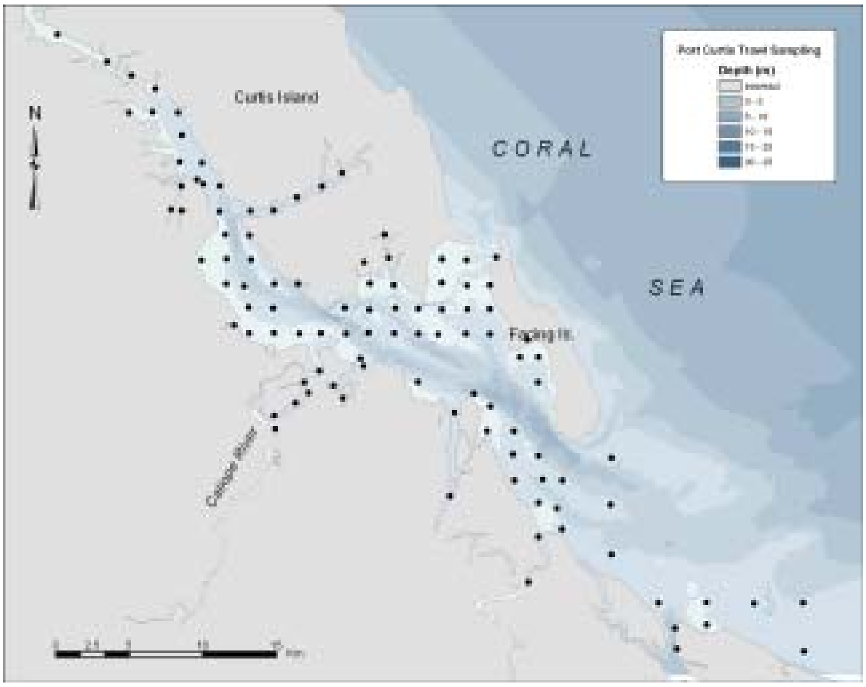


Figure 5.1. Hydrographical map of Port Curtis showing the locations of 105 sampling stations (small filled circles) surveyed for demersal fish during 2003

5.2.2 Macrobenthos

Spatial variations in the benthic community structure of Port Curtis were determined from quantitative grab samples collected one year prior to the trawl sampling program (17 July to 3 September 2002). In this study, a total of 147 stations (including 105 stations subsequently surveyed for fish) were sampled (Figure 5.2). A total of 3*0.1 m^2 replicate grabs were collected from each station with the skipper holding the vessel on site without anchor. Two sediment subsamples (70 ml and 10 ml) were retained from each replicate grab sample prior to sieving. These fractions were collected from the surface layer by scraping an open vial across the top of each sample, and were subsequently analysed for size–structure and organic carbon content (Currie & Small 2004). All grabs collected were then sieved on a 1 mm mesh screen and the fauna retained preserved in 5% formaldehyde solution. This fauna was later sorted in the laboratory to the highest possible taxonomic level (generally species) before being counted. Any seagrass collected in the grab was separated and identified, and later air-dried to provide an estimate of seagrass standing-stock.

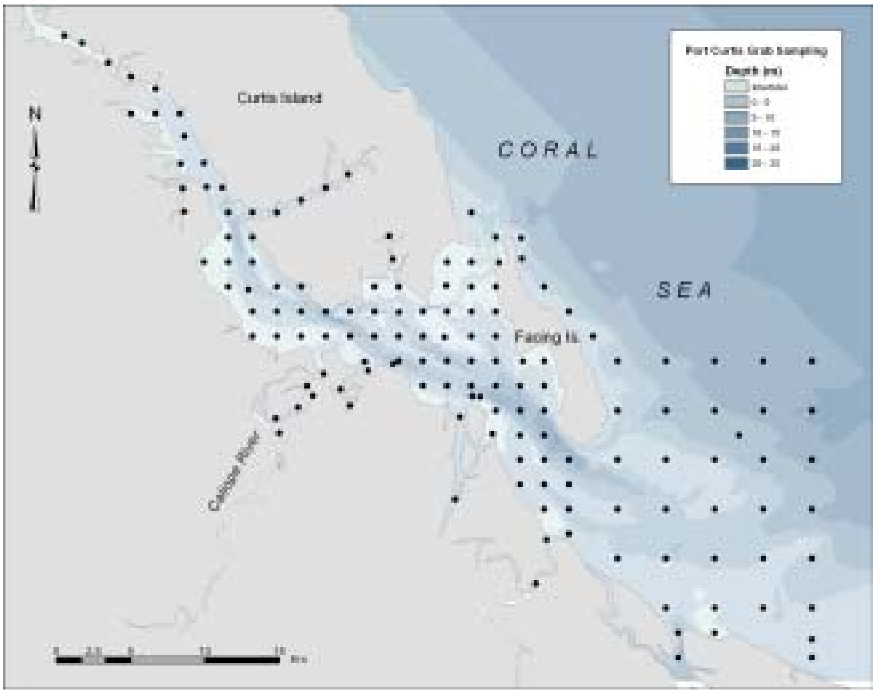


Figure 5.2. Map of Port Curtis showing the locations of 149 sampling stations (small filled circles) surveyed for macrobenthos, sediment structure and water quality during winter 2002. Of these stations, 105 (Figure 5.1) were subsequently sampled for demersal fish during winter 2003

5.2.3 Water chemistry

A comprehensive water quality survey was undertaken at the same time as the grab sampling study. Measures of water temperature, salinity, pH and dissolved oxygen content were collected from each of the 149 sampling station using an automated water quality analyser (YeoKal 611^{TM}). This instrument was preset to acquire data at 1 m intervals, and was lowered on a weighted cable through the water column at each station immediately prior to grabbing. Turbidity of the water column at each sampling station was measured using a 30 cm diameter Secchi disk. This was lowered on a metred rope beneath the research vessel until no longer visible, and the depth at which this occurred was recorded. Measures of total chlorophyll concentration were also determined for surface waters at each sampling station following the procedures detailed in ISO 10260 (1992). All water quality measurements were standardised by daily observations taken at a reference station located 500 m north-east of the entrance to Auckland Creek (Figure 5.1).

5.2.4 Data analysis

Two-way analysis of variance (ANOVA) was used to examine the effects of region and depth on fish abundance and richness. To simplify these analyses and their interpretation, sampling stations were classified into three regions that broadly reflected regional differences in winter salinity (Currie & Small 2006). In this classification, stations located west of the southernmost tip of Curtis Island were defined as 'upper estuarine'. Stations located east of Facing Island were defined as 'open coastal', while stations located between these longitudes were defined as 'central estuarine'. All stations were further classified according to depth, and three strata were defined: 'intertidal' (>1 m above MLWS), 'littoral fringe' (1 m ± MLWS), and 'subtidal' (>1 m below MLWS). Prior to these analyses, homogeneity of variance was examined by Cochran's test and heterogeneity removed by a $\log_{10}(n+1)$ transformation.

Variations in community structure of demersal fish between the 105 Port Curtis stations were examined using Bray-Curtis dissimilarity measures (Bray & Curtis 1957). This dissimilarity measure was chosen because it is not affected by joint absences, it gives more weighting to abundant than rare species, and it has consistently performed well in preserving ecological distance in a variety of simulations on different types of data (Field *et al*. 1982; Faith *et al*. 1987). Double square root transformations were applied to the data to prevent abundant species from influencing the Bray-Curtis dissimilarity measures excessively (Clarke & Green 1988; Clarke 1993).

Multivariate analyses were conducted using the computer package PRIMER (Clarke & Gorley 2001). Hierarchical agglomerative clustering was used to group sites according to their community composition, and a similarity percentage test (SIMPER) conducted to determine those species contributing most to within and between site groupings. The extent to which measured environmental variables (temperature, salinity, pH, DO, turbidity, chlorophyll, sediment structure, organic carbon, seagrass, macrobenthic abundance and macrobenthic richness) could account for observed community groupings was further tested using the BIOENV routine of Clarke and Ainsworth (1993).

A geographic information system (GIS) was employed to characterise and display spatial trends in environmental data. Physical, chemical and biological attributes for each sampling station were interpolated using an inverse distance weighting

(IDW) algorithm (Cressie 1993), and series of predictive maps were constructed. These maps were used to visualise discontinuities between homogeneous regions and highlight patterns of similarity between variables. Relationships between each environmental variable were subsequently tested using Pearson's correlation coefficients.

5.3 Results

5.3.1 Faunal characteristics

General distribution of fish

A total of 88 species of fish and 2994 individuals were collected from the 315 replicate trawl shots (3*105 stations) undertaken in Port Curtis. The ponyfish *Leiognathus equulus* was the most abundant and widespread species found during the study, and occurred at half of the stations sampled (50%), and accounted for more than a third of the total abundance (34%) (Table 5.1). Another schooling fish, the herring *Herklotsichthys castelnaui*, was also common, occurring at 36% of the sampling stations and accounting for over 16% of the total catch. All other species had restricted distributions and were found at fewer than 30% of the stations sampled, and individually contributed less than 9% to the total abundance.

Table 5.1. Total abundance (n/3.1 ha) and frequency of occurrence of fish species in beam trawl samples (3*5m*200m) collected at 105 sampling stations in Port Curtis

Species	Common name	Abundance (n/3.1ha)	Relative abundance (%)	Frequency of occurrence	Occurrence (%)
Leiognathus equulus	Common ponyfish	1030	34.40	53	50.48
Herklotsichthys castelnaui	Southern herring	489	16.33	36	34.29
Ambassis marianus	Glass perch	244	8.15	7	6.67
Megalaspis cordyla	Finny scad	121	4.04	15	14.29
Atherinomorus ogilbyi	Common hardyhead	97	3.24	17	16.19
Siganus rivulatus	Happy moments	83	2.77	9	8.57
Valamugil georgii	Fantail mullet	80	2.67	12	11.43
Pomadasys kaakan	Barred grunter	77	2.57	14	13.33
Gerres subfasciatus	Common silverbelly	65	2.17	25	23.81
Saurida undosquamis	Large-scaled grinner	65	2.17	29	27.62
Pelates quadrilineatus	Trumpeter	53	1.77	10	9.52
Acanthopagrus australis	Silver bream	47	1.57	13	12.38
Meuschenia sp. 2	Leatherjacket	42	1.40	4	3.81
Apogon fasciatus	Striped cardinalfish	30	1.00	12	11.43
Leptobrama mulleri	Steelback salmon	30	1.00	14	13.33
Liza dussumieri	Flat-tail mullet	27	0.90	8	7.62
Tripodichthys angustifrons	Yellow-fin tripod-fish	26	0.87	18	17.14
Arrhamphus sclerolepis	Snub-nosed garfish	24	0.80	7	6.67
Gerres filamentosus	Threadfin silverbelly	23	0.77	10	9.52
Pseudorhombus arsius	Large-toothed flounder	23	0.77	18	17.14
Gerres oyena	Oceanic silverbelly	20	0.67	1	0.95
Acanthopagrus sp. 2	Bream	18	0.60	1	0.95
Nematalosa come	Bony bream	18	0.60	9	8.57

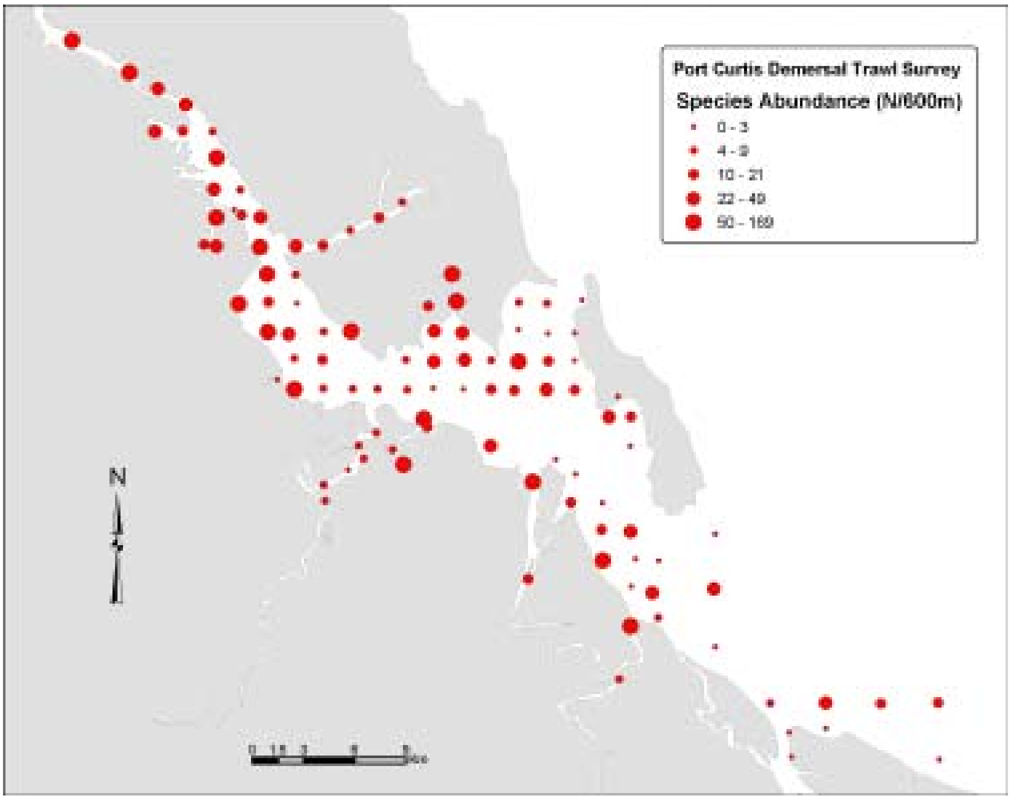
Table 5.1 (continued) Total abundance (n/3.1 ha) and frequency of occurrence of fish species in beam trawl samples (3*5m*200m) collected at 105 sampling stations in Port Curtis

Species	Common name	Abundance (n/3.1ha)	Relative abundance (%)	Frequency of occurrence	Occurrence (%)
Pelates sexlineatus	Eastern striped trumpeter	17	0.57	2	1.90
Selaroides leptolepis	Smooth-tailed trevally	15	0.50	3	2.86
Sillago maculata maculata	Winter whiting	15	0.50	10	9.52
Trepaon jarbua	Crescent perch	14	0.47	6	5.71
Drepane punctata	Sicklefish	12	0.40	8	7.62
Scomberoides commersonianus	Giant leatherskin	10	0.33	8	7.62
Thryssa aestuaria	Southern anchovy	10	0.33	2	1.90
Tetractenos hamiltoni	Common toadfish	9	0.30	7	6.67
Marilyna pleurosticta	Banded toadfish	8	0.27	5	4.76
Triacanthus brevirostris	Short-nosed tripod-fish	7	0.23	6	5.71
Upeneus tragula	Bar-tailed goatfish	7	0.23	4	3.81
Arothron manilensis	Narrow-lined toadfish	6	0.20	5	4.76
Hyporhamphus quoyi	Short-nosed garfish	6	0.20	1	0.95
Leiognathus sp. 3	Ponyfish	6	0.20	3	2.86
Leiognathus sp. 2	Ponyfish	6	0.20	3	2.86
Selenotoca multifasciata	Striped butterfish	3	0.10	1	0.95
Sillago (Parasillago) analis	Rough-scaled whiting	3	0.10	2	1.90
Sygnathidae sp.1	Pipefish	3	0.10	2	1.90
Absalom radiatus	Fringe-finned Trevally	2	0.07	1	0.95
Apogon sp. 1	Cardinalfish	2	0.07	2	1.90
Argyrosomus japonicus	Jewfish	2	0.07	2	1.90
Carangoides caeruleopinnatus	Onion-ring trevally	2	0.07	2	1.90
Chaetodon tricinctus	Three-band coralfish	2	0.07	1	0.95
Dasyatis fluviorum	Brown stingray	2	0.07	2	1.90
Hippocampus sp. 1	Seahorse	2	0.07	2	1.90
Hyperolophus translucidus	Glassy sprat	2	0.07	1	0.95
Meuschenia sp. 3	Leatherjacket	2	0.07	2	1.90
Nemipterus theodorei	Yellow-lip butterfly-bream	2	0.07	2	1.90
Parachaetodon ocellatus	Ocellate coralfish	2	0.07	1	0.95
Tylosurus crocodilus	Crocodile long-tom	2	0.07	2	1.90
Achlyopa nigra	Black sole	1	0.03	1	0.95
Auxis thazard	Frigate mackerel	1	0.03	1	0.95
Choerodon cephalotes	Purple tuskfish	1	0.03	1	0.95
Dasyatis kuhlii	Bluespot stingray	1	0.03	1	0.95
Diodor nichthemerus	Porcupine fish	1	0.03	1	0.95
Eleutheronema tetradactylum	Blue threadfin salmon	1	0.03	1	0.95
Epinephelus coioides	Orange-spotted cod	1	0.03	1	0.95
Gymnura australis	Rat-tailed ray	1	0.03	1	0.95
Himantura uarnak	Long-tailed ray	1	0.03	1	0.95
Hyporhamphus australis	Sea garfish	1	0.03	1	0.95
Meuschenia sp. 1	Leatherjacket	1	0.03	1	0.95
Meuschenia sp. 4	Leatherjacket	1	0.03	1	0.95
Muraenesox cinevus	Pike eel	1	0.03	1	0.95
Pentapodus paradiseus	Blue-faced whiptail	1	0.03	1	0.95
Periopthalmus koelreuteri	Mud-skipper	1	0.03	1	0.95
Platycephalus arenarius	Sand flathead	1	0.03	1	0.95
Pseudorhombus argent	Flounder	1	0.03	1	0.95
Scomberomorus queenslandicus	School mackerel	1	0.03	1	0.95
Scorpaenopsis sp.	Scorpionfish	1	0.03	1	0.95
Ulua mentalis	Cale trevally	1	0.03	1	0.95
Total		**2994**		**105**	

Regional and depth-related differences in fish abundance and richness

Analysis of variance showed that there were no significant differences in the number of species found between the three regions of the port (upper estuarine, central estuarine and open coastal) (ANOVA: $F_{2,96} = 1.81$, $p = 0.169$), although plots of mean richness suggest a general decline in diversity towards open coastal waters of the estuary (Figures 5.3 and 5.4).

a)



b)

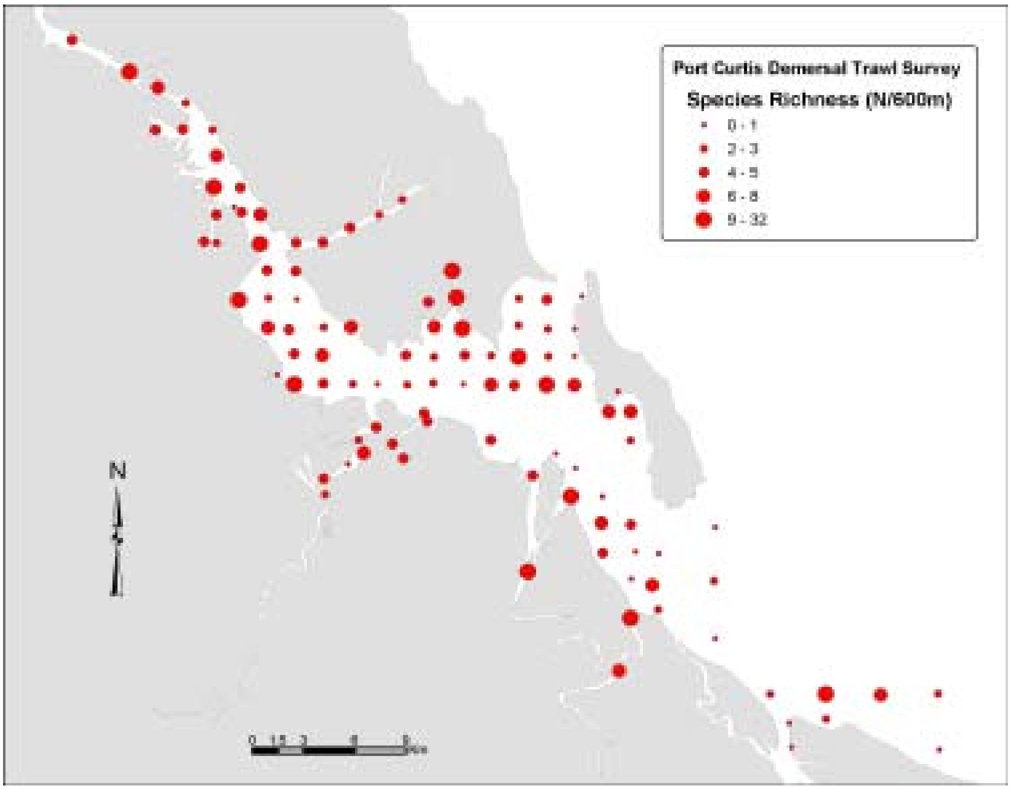
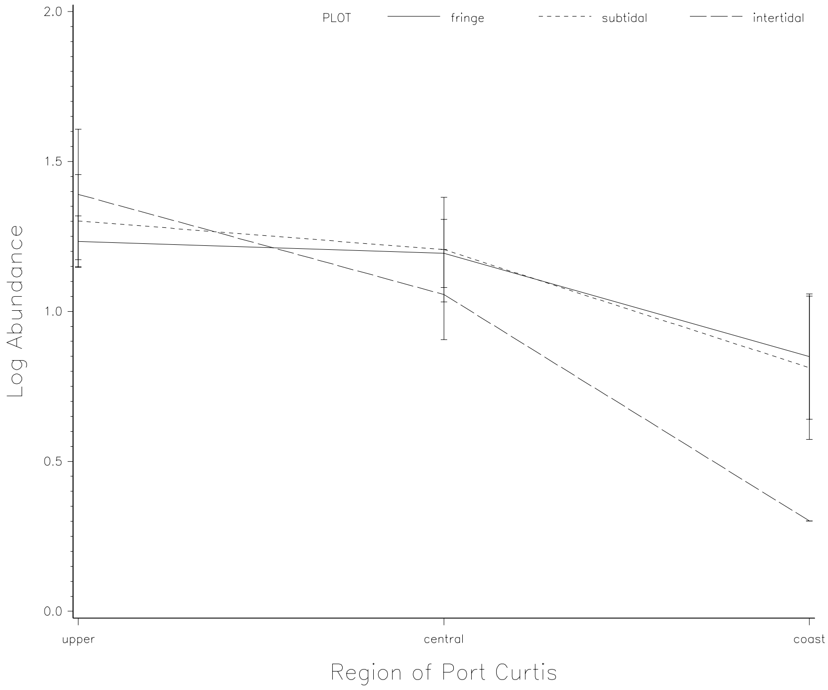


Figure 5.3. Map of Port Curtis showing a) total species abundance and b) total species richness of demersal fish collected from 3 replicate beam trawl samples (200 m length) at 105 sampling stations

a)



b)

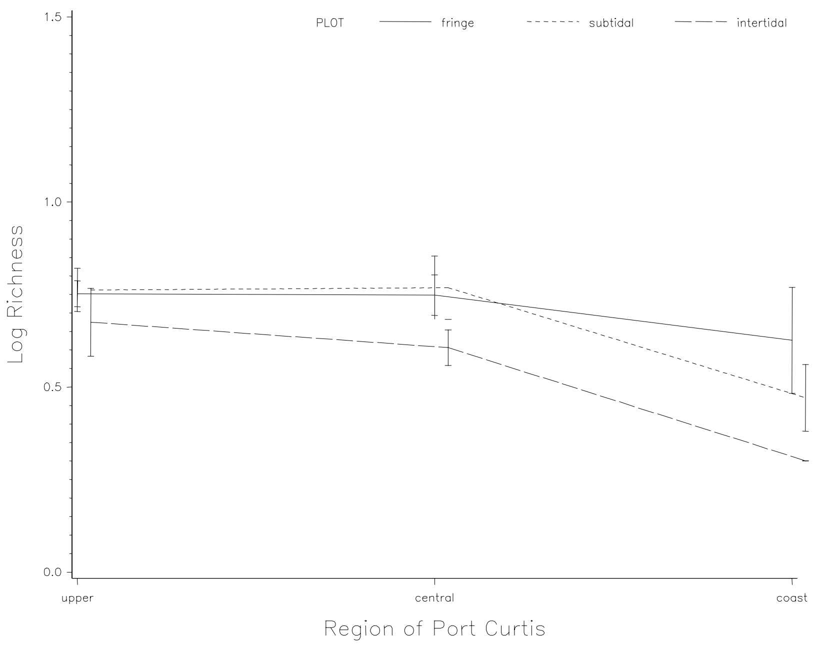


Figure 5.4. Plots of a) mean species abundance and b) mean species richness of demersal fish in Port Curtis. Means (± s.e.) are derived from replicate 200 m trawl shots at 105 sampling stations. These stations have been classified here according depth (intertidal, littoral fringe and subtidal) and regional location (upper estuarine, central estuarine an open coastal)

Community structure

Five station groupings were separated by cluster analysis of species abundance data (Figure 5.5), and their corresponding distributions plotted (Figure 5.6). These groupings closely reflected differences in depth and habitat type within Port Curtis. Group I was intimately associated with the seagrass beds (Pelican Banks, South Trees and Tannum).

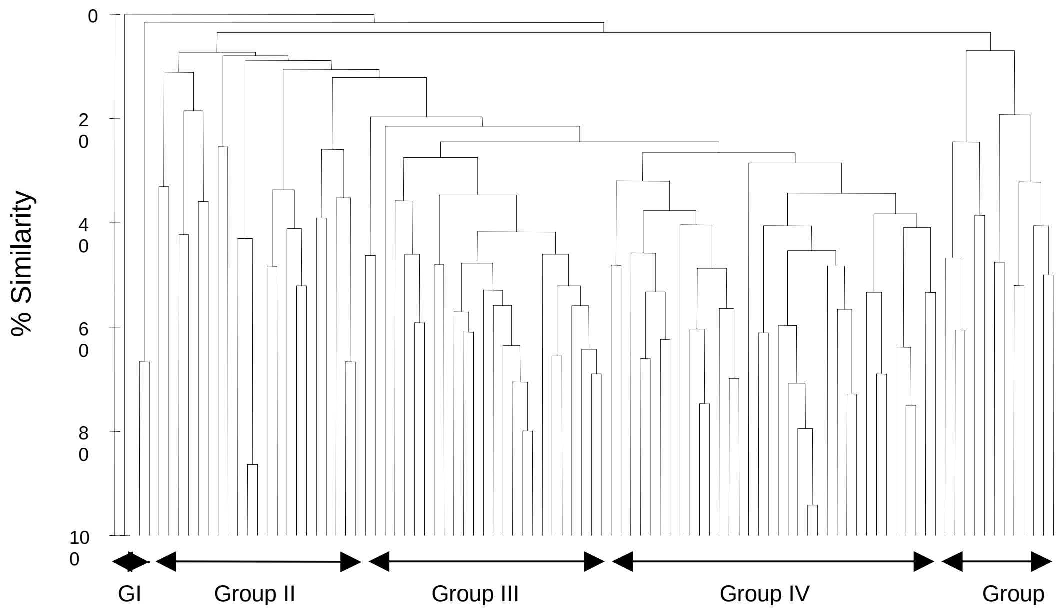


Figure 5.5. Dendrogram of Bray-Curtis similarities derived from demersal fish abundance data (double square root transformed), with five station groups identified

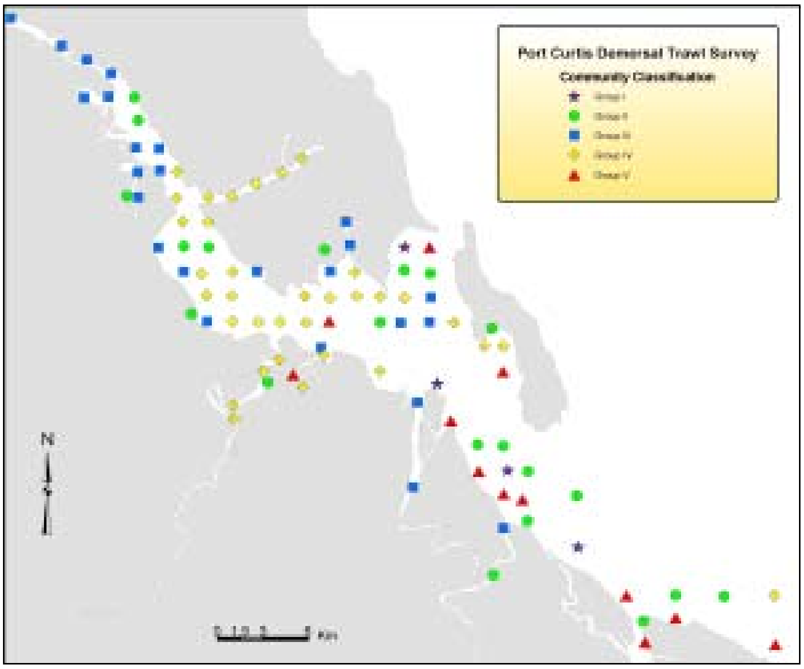


Figure 5.6. Map showing the locations of 105 demersal fish sampling stations and their classification into five groups following cluster analysis of species abundance data

SIMPER analysis was undertaken to assess which species contributed most to similarities within and differences between the five station groupings. Abundances of the 14 species contributing ≥ 5% to within-group similarity or between-group dissimilarity for at least one of the five groupings are given in Table 5.2. The average within-group similarity ranged from 9–35%, and between-group dissimilarity ranged from 76–99%. Two station groups (I and V) were independently characterised by small subsets of species with restricted distributions. The other three groups (II, III and IV) had generally higher diversity and were differentiated by varying proportions of co-occurring species.

The PRIMER routine BIOENV was used to assess the correspondence and significance of environmental data to the five station groupings. The best fit was with turbidity ($\rho_w = 0.22$), which in combination with depth, infaunal richness and seagrass biomass gave a best fit of $\rho_w = 0.34$. The remaining variables (sediment, pH, salinity, temperature, dissolved oxygen, organic carbon, chlorophyll and infaunal abundance) were apparently unrelated to any pattern in station groupings ($\rho_w < 0.14$).

Table 5.2. Abundance (n/3000 m^2) of demersal fish in five regions of Port Curtis identified from hierarchical classification of species/abundance data

Species listed were identified from SIMPER analyses as contributing ≥ 5% to the similarity within and dissimilarity between regional groupings. Those species indicative of each regional grouping (contributing ≥ 5% to the total similarity within a group) are highlighted in bold. Species are ranked in order of decreasing abundance across all stations.

Species	Common name	Station grouping				
		1	2	3	4	5
Leiognathus decora	Common ponyfish			**28.28**	**9.47**	
Herklotsichthys castelnaui	Southern herring		**1.43**	**12.40**	4.38	
Ambassis marianus	Glass perch			7.68	1.53	
Megalaspis cordyla	Finny scad		0.88	0.32	2.53	
Siganus rivulatus	Happy moments			0.08		**6.75**
Saurida undosquamis	Large-scaled grinner		**0.29**	0.80	**1.15**	
Apogon fasciatus	Striped cardinalfish			0.36	0.50	**0.33**
Liza dussumieri	Flat-tailed mullet		**0.62**	0.24		
Tripodichthys angustifrons	Yellow-fin tripod fish		**0.29**	0.24	0.38	
Pseudorhombus arsius	Large-toothed flounder		**0.29**	0.12	0.38	
Sillago maculata maculata	Winter Whiting				0.21	**0.67**
Scomberoides commersonianus	Giant leatherskin	**0.05**		0.24		
Upeneus tragula	Bar-tailed goatfish					**0.50**
Hippocampus sp. 1	Seahorse	**0.50**				

5.4 Discussion

Fish assemblages of Port Curtis are diverse, with 88 species of fish recorded. The dominance of the fish fauna by two small schooling species is typical of the fauna recorded in similar studies elsewhere in Queensland (Blaber *et al*. 1989; Thomas & Connolly 2001). In this case, the dominant species were ponyfish (*Leiognathus equulus*) and herring (*Herklotsichthys castelnaui*), both species common in inshore coastal waters elsewhere in tropical and subtropical waters of Australia.

Fish assemblages change gradually from the more estuarine Narrows to open coastal waters. This is essentially an estuarine gradient, known to occur elsewhere in Australia (West & King 1996) but not previously reported in subtropical waters. The influence of salinity, *per se*, is not strong. If anything, turbidity is more important, since the correlation between fish and turbidity was the strongest measurable relationship. Alternatively, the degree of exposure to open coastal waters might itself be influencing fish distributions (Connolly 1994), but this is difficult to separate from turbidity since open coastal waters are typically less turbid in any case.

In addition to the estuarine gradient, five assemblage types were defined, reflecting differences in habitat type and depth. This comes as no surprise since both habitat and depth have been shown to be important in determining fish assemblages in several other places (Jackson *et al*. 2001). The group of fish associated with seagrass is of particular note, since it confirms in Port Curtis the importance of this habitat type, as reported elsewhere in Australia (Connolly 1994) and internationally (Jackson *et al*. 2001).

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Chapter 6. Trophic importance of intertidal wetland habitats in Port Curtis

Rod M. Connolly

6.1 Introduction

Port Curtis is a large subtropical embayment with very extensive intertidal and shallow subtidal wetland habitats (see Chapter 2). Individuals of economically important fish and crustacean species occur in all of these habitats. In terms of habitat protection and conservation, previous studies have focused mainly on where the animals occur (i.e. they examine spatial patterns in animal assemblages). This has been true for studies elsewhere along the Queensland coast (e.g. Blaber & Blaber 1980; Thomas & Connolly 2001), and is also the approach taken in Chapter 5 in this report. While this inventory work is essential, the importance of a habitat to fisheries production does not stop with the animals actually found in a particular habitat. Water can act as a vector for the movement of energy, and autotrophic (plant or algal) production may be transferred to adjacent wetland habitats. Autotrophic production in one habitat may therefore form the base of food webs that support fisheries species in spatially separated habitats (Melville & Connolly 2003, 2005).

The principle aim of this work was to conceptualise the flow of energy from the various wetland habitats to fisheries species, especially those occurring over shallow mudflats lacking conspicuous vegetation. The autotrophic source(s) supporting food webs leading to animal production on the mudflats might be either *in situ* microalgae or material transported from adjacent habitats dominated by macrophytes. This conceptualisation draws on previous work by the author (Connolly & Guest in press) and the main points of that work are mentioned again here to support the conceptualisation.

A secondary aim was to test directly whether small fish feed on the saltflats that form extensive, high intertidal habitat in Port Curtis. New field and laboratory evidence collected for this test is reported here for the first time.

6.2 Methods

6.2.1 Energy pathways

Stable isotope analysis was used to determine the trophic (feeding) pathways in estuarine food webs in Port Curtis. Stable isotope analysis traces energy and nutrients through food webs, and is the best method for separating the importance of different vegetation types (habitats) to animals. This form of chemical tracing works whether the animals of interest are herbivores, detritivores or carnivores, since the stable isotope tracer can be followed through multiple trophic levels. Specifically, the carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$) isotope values of nine fish and four crustacean species were measured (Table 6.1), as were eight autotroph (primary producer) taxa, as follows: in adjacent habitats, *Zostera* seagrass, *Halophila* seagrass, mangroves, saltmarsh succulents, saltmarsh grass and algal mats; on mudflats, microphytobenthos; and in the water column, particulate organic matter. Microphytobenthos (MPB) is mostly microscopic algae known as benthic microalgae, living on and in mud, but also include cyanobacteria. Particulate organic matter (POM) includes phytoplankton and detritus in the water column. Sampling of consumers and autotrophs was done at three locations in Port Curtis (Facing Island, Calliope River mouth and The Narrows), but locations were pooled prior to analysis (see Connolly & Guest in press). In selecting the animal species to be analysed, the aim was to use fish and crustacean species harvested recreationally and commercially, as well as a small number of species not harvested but very abundant in Port Curtis. Further detailed descriptions of the collection and processing of samples are provided in Connolly and Guest (in press).

Stable isotope ratios of animal tissue are well suited to calculating the likelihood of contribution of different autotrophic sources to food webs. Different autotrophs (plants and algae) have different stable isotope ratios, and these are more or less faithfully propagated through different trophic levels in the food web. Modelling of animal isotope ratios therefore allows us to determine which plants (and therefore which habitats) are likely to be contributing to food webs supporting fisheries production.

6.2.2 Euclidean modelling

Autotrophs were pooled into eight taxa: mangroves, *Zostera* seagrass, *Halophila* seagrass, POM, MPB, algal mats, the C_3 saltmarsh succulents and the C_4 saltmarsh grass. Mean $\delta^{13}C$ and $\delta^{15}N$ values were calculated for each animal and autotroph taxon. Only animal species for which more than one specimen was obtained were modelled (six fish and three crustacean species). Using the $\delta^{13}C$ and $\delta^{15}N$ values as Cartesian coordinates (after correction for fractionation of three units per trophic level for $\delta^{15}N$), Euclidean distances (E) between fish values and each of the autotroph categories were calculated according to:

$$E = [(\delta^{13}C_{autotroph} - \delta^{13}C_{fish})^2 + (\delta^{15}N_{autotroph} - \delta^{15}N_{fish})^2]^{0.5}$$

Variances were calculated about these Euclidean distances as follows:

$$s^2 = a \times s^2 (\delta^{13}C_{autotroph}) + a \times s^2 (\delta^{13}C_{fish}) + b \times s^2 (\delta^{15}N_{autotroph}) + b \times s^2 (\delta^{15}N_{fish})$$

where s^2 = variance, $a = ((\delta^{13}C_{autotroph} - \delta^{13}C_{fish}) / \text{distance})^2$ and $b = ((\delta^{15}N_{autotroph} - \delta^{15}N_{fish}) / \text{distance})^2$.

A small Euclidean distance between a fish and an autotroph indicates a large putative dietary contribution, so distances were inverted to make the measure more intuitive. The inverted distance for each autotroph was then calculated as a percentage of the total of the inverted distances for all autotrophs for a particular fish species.

The focus of the work was to conceptualise energy pathways into a diagrammatic framework consistent with previous outputs from the Coastal CRC.

6.2.3 Feeding by fish on saltflats

Juvenile fish were collected from Boat Creek, near Fisherman's Landing wharves in the lower part of The Narrows. Fish were collected at two phases of an inundation cycle: pre-inundation which was done prior to marsh inundation, and post-inundation as water was flowing off the edge of the marsh.

Pre-inundation sampling of fish and crustaceans was done using a combination of a large (10 x 2 m x 10 mm mesh) and small (5 x 2 m x 1 mm mesh) seine net. The post-inundation sampling was done using two square hoop fyke nets (0.7 x 0.7 m square, 10 m wings). The intention of these methods was to catch fish moving onto and off the saltflats, not to quantify fish abundances.

The number of fish caught pre-inundation was low, so the two most common species were selected for analysis: fantail mullet (*Valamugil georgii*), 16 pre, 212 post, mean fork length 65 mm; sand whiting (*Sillago ciliata*), 4 pre, 159 post, mean fork length 43 mm). Stomach content analysis involved the quantification of stomach fullness (using an arbitrary 0–5 scale where 0 is empty and 5 is full) and the average estimated abundance (%) of each prey type (Berg 1979).

Table 6.1. Fish and crustacean species analysed by Connolly and Guest (in press) and trophic levels used for correction of fractionation for each species (trophic-level literature sources in Connolly & Guest in press)

Trophic levels are appropriate for the stage of life collected during this study for each species. Half-trophic levels result from a species taking some prey at one level and other prey at other levels.

Species	Common name	Trophic level(s) above autotrophs
Fish		
Acanthopagrus australis	Yellowfin bream	2.0
Arrhamphus sclerolepis	Snub-nosed garfish	1.5
Drepane punctata	Sicklefish	2.5
Gerres subfasciatus	Common silverbiddy	2.0
Herklotsichthys castelnaui	Southern herring	2.0
Hyporhamphus quoyi	Short-nosed garfish	1.5
Leiognathus equulus	Common ponyfish	2.0
Sillago ciliata	Sand whiting	2.0
Valamugil georgii	Fantail mullet	1.0
Crustaceans		
Fenneropenaeus merguiensis	Banana prawn	1.5
Oratosquilla stephensoni	Stephenson's mantis prawn	2.5
Penaeus esculentus	Tiger prawn	1.5
Scylla serrata	Mud crab	2.0

6.3 Results and discussion

6.3.1 Energy pathways

The contribution of each autotroph to fish species was modelled using a Euclidean mixing model. Fish $\delta^{13}C$ values lay exclusively in the enriched half of the range for autotrophs, indicating very minor contributions from depleted autotrophs (mangroves, saltmarsh succulents) (Figure 6.1). Seagrass (mainly *Zostera*) was in the top three potential contributors for all fish species (Table 6.2). For crustaceans, *Halophila* seagrass had the highest potential contribution. Organic matter from seagrass beds appears to be an important source for animals on adjacent unvegetated mudflats, either through outwelling of particular organic matter or via a series of predator–prey interactions (trophic relay). Saltmarsh grass (*Sporobolus*) also had high contributions for many species. Although macrophyte production in adjacent habitats was the dominant source of nutrition for the suite of animals over unvegetated mudflats, *in situ* microphytobenthos had a high contribution to 50% of fish and one of the crustacean species (mud crab, *Scylla serrata*), and particulate organic matter, including phytoplankton, was a likely contributor to several other species.

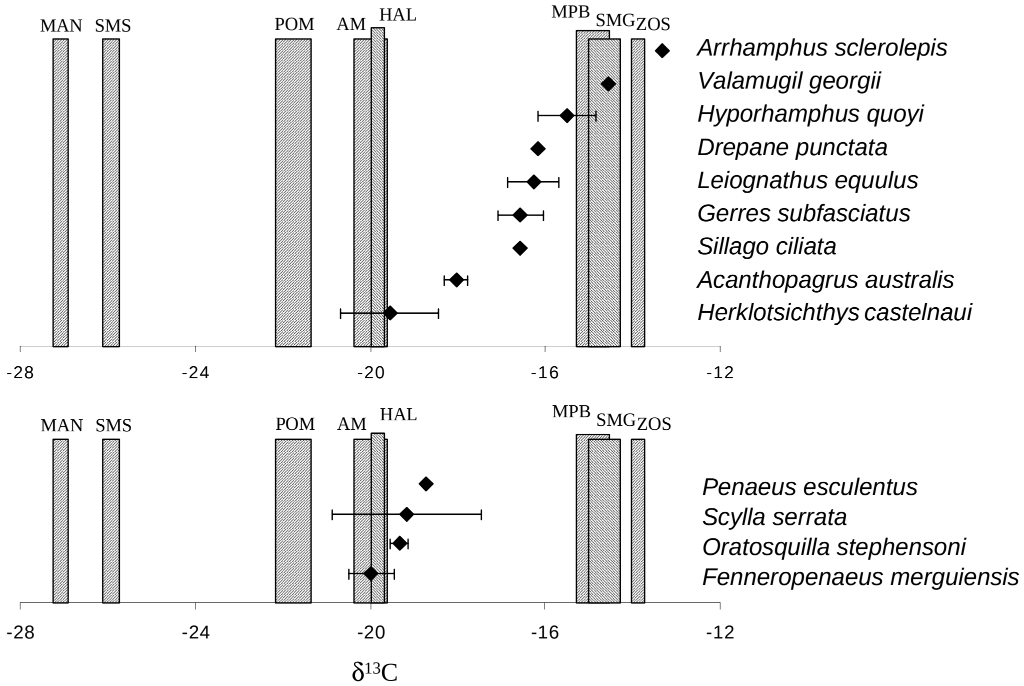

Figure 6.1. Mean $\delta^{13}C$ (‰) values of fish and crustaceans overlaid on autotroph values (after Connolly & Guest in press)

For animals, values are shown as a mean (diamond) ± s.e. For autotrophs, mean ± s.e. is shown by the position and width of columns, using the symbols: AM = Algal mat; ZOS = Zostera; MAN =mangroves; MPB = microphytobenthos; POM = particulate organic matter; SMG saltmarsh grass; SMS = saltmarsh succulents; HAL = Halophila.

Table 6.2. Summary of Euclidean mixing model results (after Connolly & Guest in press)

Results exclude two fish and one crustacean species for which too few specimens were collected. Values are number of fish species out of six and number of crustacean species out of three for which the contribution of an autotroph is important, ranked by contribution (1, 2 or 3). Percent values in final column represent combined rankings. For example, *Zostera capricorni* was the first ranked contributor for two out of six fish species, and was in the top three rankings for four out of six species (i.e. 67%).

Autotroph	Source	Rank 1	Rank 2	Rank 3	Total	%
Fish (6 species)						
Saltmarsh grass	outwelled	3	1	–	4	67
Zostera capricorni	outwelled	2	1	1	4	67
Halophila ovalis	outwelled	2	1	–	3	50
MPB	*in situ*	–	1	2	3	50
Algal mat	outwelled	–	2	–	2	33
POM	*in situ*, outwelled	–	1	1	2	33
Mangrove	outwelled	–	–	–	–	–
Saltmarsh succulent	outwelled	–	–	–	–	–
Crustaceans (3 species)						
Halophila ovalis	outwelled	2	–	1	3	100
Algal mat	outwelled	–	1	1	2	67
POM	*in situ*, outwelled	–	1	1	2	67
MPB	*in situ*	1	–	–	1	33
Saltmarsh grass	outwelled	–	1	–	1	33
Mangrove	outwelled	–	–	–	–	–
Saltmarsh succulent	outwelled	–	–	–	–	–
Zostera capricorni	outwelled	–	–	–	–	–

6.3.2 Conceptual model

The conceptual model (Figure 6.2) depicts our current understanding of energy flows from different wetland habitats to food webs supporting fisheries species in Port Curtis. Landscape features include industrial, port (shipping), urban and agricultural activities. Although the focus of the project is on the industrial and port impacts on Port Curtis, there are nevertheless a major regional city and sizeable agricultural areas on the western side of Port Curtis.

Energy flows from the various wetland habitats to fisheries food webs are shown using arrows, where the width of the arrow indicates the extent of contribution. For fish species, the major contributing habitats are seagrass (predominantly *Zostera*), and to a lesser extent microphytobenthos (MPB), with a very minor contribution from mangrove organic matter. For crustaceans (mud crabs and banana prawns), the seagrass *Halophila* replaces *Zostera* as the most likely source of carbon to the food webs. As for fish, MPB also contribute and there is a minor contribution from mangroves.

Stable isotope analysis is unable to separate contributions from autotrophs that have the same isotope ratios, as is the case with *Zostera* seagrass and the saltmarsh grass *Sporobolus*. The high putative contributions assigned to *Sporobolus* for fish and crustaceans in the isotopic modelling might therefore be misleading. Saltmarsh grass grows high in the intertidal zone, along the edges of the extensive saltflats landward of the mangroves, and is probably not a major contributor to carbon and nutrient budgets in the Port. This lack of productivity of saltmarsh grass, along with evidence from Moreton Bay studies showing that seagrass rather than saltmarsh grass drives food webs, suggest that the contribution from saltmarsh grass remains unsubstantiated, and further work with other tracers is warranted.

The conceptual understanding is presented here as a three-dimensional diagram. A decision was made that the conceptual diagrams in Chapter 7 would be better able to show processes using two-dimensional diagrams, and so the trophic diagram in that chapter has similar content but a slightly different look.

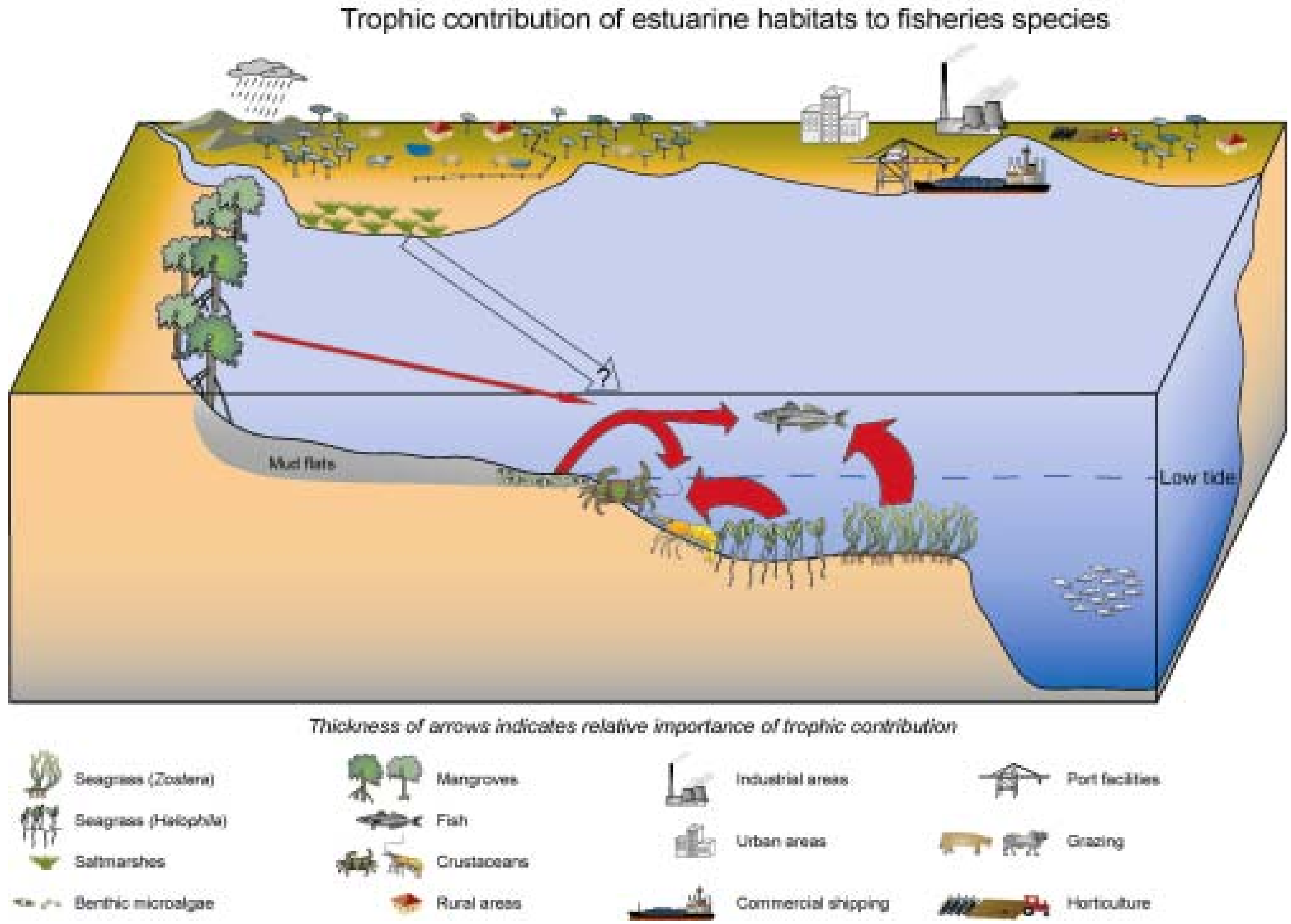

Figure 6.2. Conceptual model of trophic contributions to fisheries species over mudflats in Port Curtis. Saltflats landward of the saltmarsh grass not shown, but make little contribution to food webs. Saltmarsh grass shown at water edge also occurs behind mangroves fringing the waterways

6.3.3 Feeding by fish on saltflats

Stomach fullness for fantail mullet was significantly greater post- than pre-entering the saltflat (Figure 6.3). The proportion of sediment and algae decreased after being on the saltflat, and a range of invertebrates, particularly benthic crustaceans and polychaete worms, were ingested that had not been in fish stomachs pre-entering the saltflat (Figure 6.4).

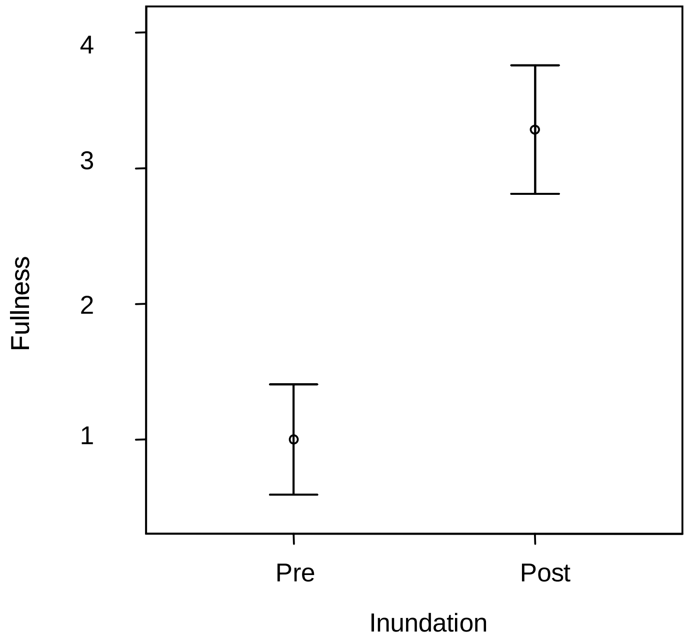

Fig 6.3. Stomach fullness (mean ± s.e.) of fantail mullet collected pre- and post-inundation of the saltflat. ANOVA: $p<0.001$

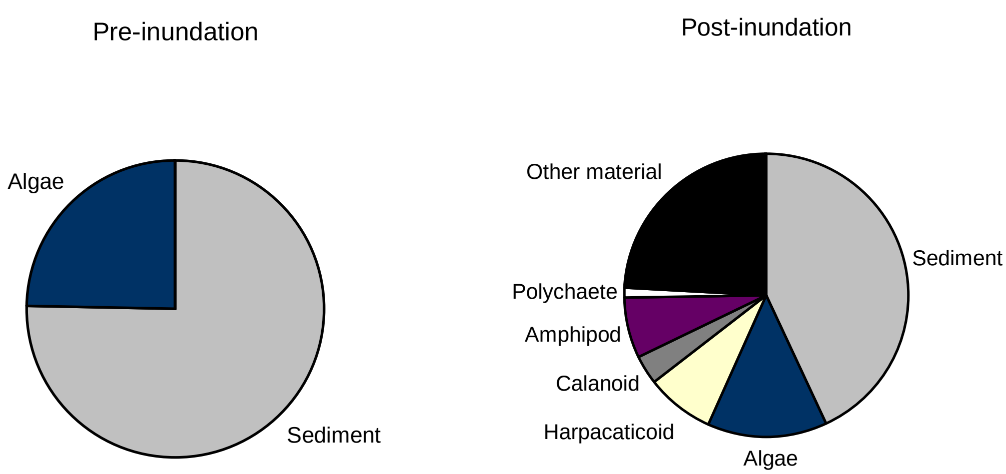

Fig 6.4. Stomach content composition of fantail mullet collected pre- and post-inundation of the saltflat. 'Calanoid' and 'Harpacticoid' refer to types of copepods (microscopic crustaceans)

These juvenile mullet certainly fed on the saltflat while it was inundated, and this is the first record we know of that describes feeding by fish on an unvegetated saltflat in subtropical or tropical waters. Several studies have demonstrated feeding on vegetated saltmarsh by comparing stomach fullness and prey composition of fish entering and leaving marsh habitat. Studies in the USA (Nemerson & Able 2004) and Europe (Laffaille *et al*. 2002) have detected higher stomach fullness after fish visit marshes, and have shown that a range of invertebrates, both marine (polychaete worms and amphipods) and terrestrial (insects), are ingested. Australian marshes are higher in the intertidal than most of the northern hemisphere marshes studied, and are therefore inundated less frequently and for shorter periods (Connolly 1999).

The feeding behaviour of fish on Australian saltmarshes should not therefore be assumed to be the same as on the better studied marshes of the northern hemisphere. Although fish use vegetated marshes in southern Queensland extensively, this has not been shown for the extensive saltflats of Port Curtis. Our own attempts in Port Curtis to estimate densities of fish on saltflats during inundation failed because densities were so low. The number of fish visiting the saltflats cannot be considered high, but clearly those fish that do visit the saltflats gain access to food resources. Further work is required to establish the spatial (e.g. less sheltered sites on outer island of Port Curtis) and temporal (seasons other than the late spring period of the current study) generality of the finding for mullet.

Whiting stomach fullness tended to be lower after visiting the saltflat, although this difference was not significant (Figure 6.5). This indicates little feeding on the saltflats. Stomach contents were dominated by crustaceans and polychaetes, and the relative proportions of different types of crustaceans varied pre- and post-inundation (Figure 6.6), indicating either minor feeding activity or differential rates of stomach passage (digestion) for different crustacean types. Whiting apparently made little use of the saltflat for feeding, although the conclusion for this species is drawn from a small sample size and, again, further work is required to establish the generality of the result.

The overall conclusion from the stomach content work is that saltflats in Port Curtis are visited by relatively small numbers of fish, and that the habitat can be important in their feeding strategy but that this importance varies among fish species.

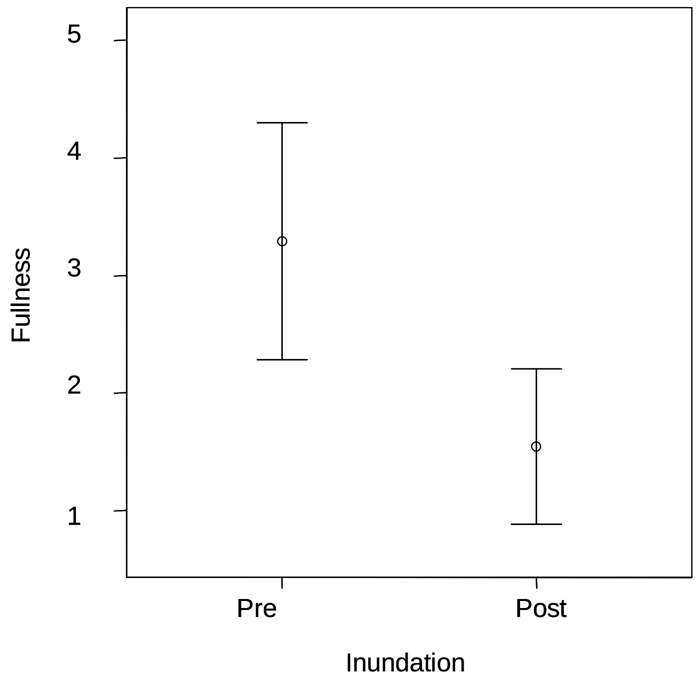

Fig 6.5. Stomach fullness (mean ± s.e.) of whiting collected pre- and post-inundation of the saltflat. ANOVA: not significant, p = 0.195

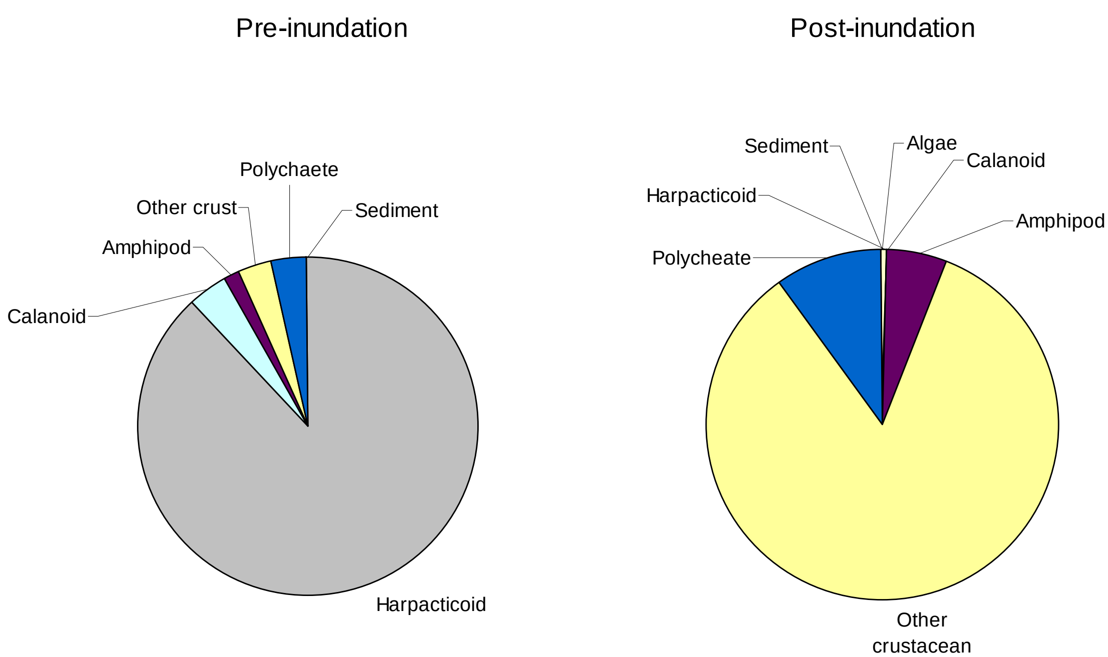

Figure 6.6. Stomach content composition of whiting collected pre- and post- inundation of the saltflat. 'Calanoid' and 'Harpacticoid' refer to types of copepods (microscopic crustaceans)

6.4 References

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Chapter 7. Conceptual modelling of the intertidal wetlands of Port Curtis

Peter J. Stratford and Rod M. Connolly

7.1 Introduction

The initiative for the development of conceptual models is to bring together the information and knowledge currently available for the Port Curtis region. The models present current knowledge in a simplified and informative manner that can be used and understood by the scientific community, resource managers, other stakeholders and the general public.

Several questions were encountered during the development of the conceptual models presented in this report. Firstly, how can the models provide the most useful information to resource managers and stakeholders? Secondly, how can the information be displayed in a manner able to be understood with minimal interpretive text? Thirdly, the available information for the Port Curtis system is highly variable, with a high degree of diversity in areas, habitats and ecosystems. Therefore, how can this diverse information be compressed and displayed in a small number of models? Finally, how can the models best take account of the considerable uncertainty about the degree of impact or influence exerted on the system by the major industrialisation that has occurred since the 1960s?

The models represented here have been designed to illustrate several of the most significant processes and pressures acting on the Port Curtis system. The key issues were selected from a broader list through an expert elicitation process. Eight scientists who have expertise in wetland science (authors of this report plus connected Coastal CRC staff) were asked to rank the list of processes and pressures in order of importance based on their own knowledge of the system and taking into account the issues raised at stakeholder meetings. Table 7.1 lists the processes and pressures that were identified by stakeholders at these meetings as being of concern in Port Curtis. From this list, the overall ranking was established and a shortlist of the key processes and pressures was developed. Five key processes acting on the system and five key pressures were selected in a similar manner. Conceptual models were then developed around these key processes and pressures and are presented in this report.

The models developed as part of this report have intentionally been kept to a generic format, with no single identifying feature for any one area. This is partly due to the lack of comprehensive data for any particular area, and partly due to the anticipation that these models may have a broader application to other industrial ports, both in Australia and internationally. As such, data from all sources of information have been combined to provide generic models of the key processes and pressures occurring within Port Curtis.

Table 7.1. Initial list of processes and pressures raised by stakeholders and scientists, deemed to be important for the Port Curtis intertidal wetland area

Processes	Pressures
Drought	Access to foreshore and tributaries
Evaporative potential	Acid sulfate exposure
Freshwater flow (over land and streams)	Boat strike/wash
Lyngbya (toxic)	Climate change
Nutrients	Coastal morphology
Porewater salinity	Contaminants and novel chemicals
Resuspension	Dams, weirs, barriers (in rivers and streams)
Saltmarsh/flat extent	Dredging
Sedimentation	Fishing
Solar irradiation	Hinterland land use
Storm/cyclone events	Industrialisation and urbanisation
Tides (hydrodynamic)	Intertidal infill
Trichodesmium	Shipping/construction activity
Trophic links	Stream diversion
	Thermal pollution
	Urban stormwater and sewage

All models have been developed as a two-dimensional cross-section of the intertidal zone, as it is considered that this approach highlights the actual processes and pressures in a more focused way. A key to the important features within each model is provided, along with annotated descriptions of processes, including relevant literature. Where sufficient knowledge of the Port Curtis region is lacking, inferences have been made from the literature. These areas are highlighted with dotted or solid question marks, indicating some or no knowledge, respectively. The width of the arrows on each diagram indicates the relative magnitude of a process.

Models for different processes and pressures may focus on a particular area of the intertidal zone and this is indicated by the red rectangle in the plan diagram, top right corner. The symbols used for these conceptual models are courtesy of the Integration and Application Network, Center for Environmental Science, University of Maryland. Some symbols have been modified or added to better reflect local conditions. A glossary for some of the terms used in the diagrams and text is included in Section 7.5.

7.2 Processes acting on and driving the system

Numerous physical and biological processes are known to drive wetland systems. In the development of the conceptual model it was deemed impractical to attempt to incorporate all of these processes into one small diagram. Consequently, five processes that have been identified as the key processes sustaining the Port Curtis intertidal wetlands are summarised in this model. Further detail of each process is provided in the models shown in Diagrams 7.2–7.6.

1. **Tides** – Port Curtis has a tidal range in excess of four metres, with approximately one-third of the Port area being intertidal zone (Currie & Small 2006). See Diagram 7.2.
2. **Freshwater flow** – Runoff from the surrounding environment is an important aspect of many wetland ecosystems. Port Curtis generally experiences moderate rainfall, with a mean annual rainfall of approximately 918 mm (BOM 2006). Anecdotal evidence suggests that freshwater input is an important factor in the functioning of wetland systems. The manner in which the water reaches the intertidal zone may be equally important. See Diagram 7.3.
3. **Trophic links** – Fishing is an extremely important recreational and commercial activity in Port Curtis. This model (see Diagram 7.4) emphasises the energy pathways that sustain fisheries production in the Port.
4. **Severe storm/cyclone events** – Severe storm events are relatively rare in Port Curtis, although it has been suggested that such events occur frequently enough to prevent stable biotic communities (i.e. equilibrium) being reached. Some of the more influential processes occurring in relation to storm events are illustrated in Diagram 7.5.
5. **Drought conditions** – Like much of Australia, Port Curtis experiences low but highly variable rainfall. Occasionally, below-average rainfall can lead to drought conditions, with current weather patterns indicating a trend to more frequent droughts in the future. The main implications of drought for intertidal communities are shown in Diagram 7.6.

7.2.1 Processes overview diagram

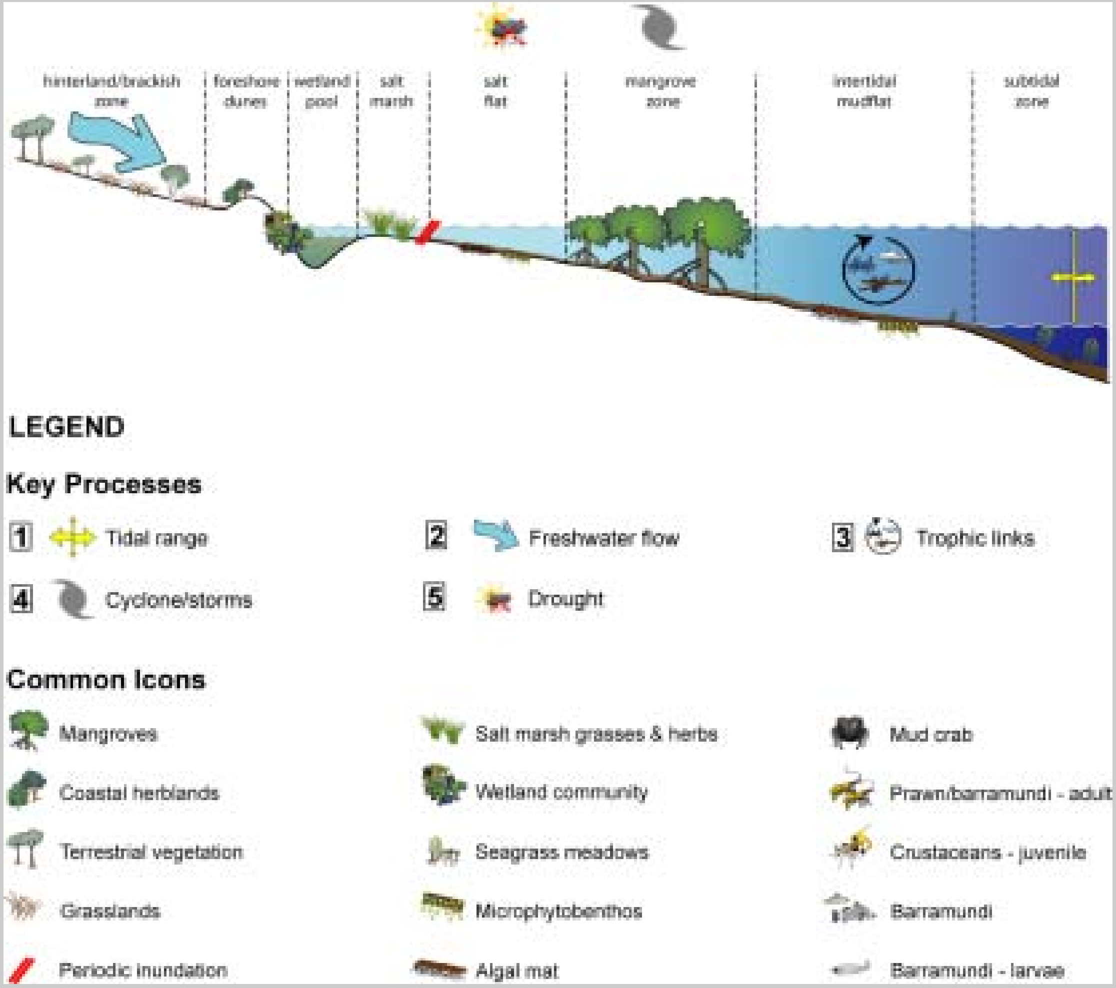

Diagram 7.1. Overview of processes acting on and driving wetland systems

The above diagram represents an overview of the key processes identified as most important for the Port Curtis intertidal wetlands: tides, freshwater, trophic links, storm events and droughts.

Each of the following conceptual diagrams includes a key to the icons particular to that diagram; however the legend in the overview model above describes the main icons, such as mangroves, saltmarsh grasses, and mud crabs, that are common to all the process diagrams and which are not shown in each individual icon key.

Each diagram is also accompanied by numbered text to describe the process depicted in that model. In addition, the width of the arrows on each diagram indicates the relative magnitude of a process.

7.2.2 Tides (Diagram 7.2)

Tidal flows are an extremely important water movement process in the wetlands of Port Curtis, particularly given that the amplitude of the tides is in excess of 4 m (Herzfeld *et al.* 2004). The elongated shape of Port Curtis and the central shipping channel impose additional implications for the intertidal wetlands, mainly through additional sedimentation, increased turbidity and higher current velocities. Diagram 7.2 demonstrates the processes associated with tidal flows within Port Curtis.

1. Flood and ebb tides provide vast areas of habitat with a huge quantity of resources available for food and shelter. Approximately one third of Port Curtis harbour is classified as intertidal zone (Currie & Small 2006). Flood tides generally provide resources and habitat extension for aquatic species, including dugong and sea turtles, while the ebb tide provides resources for non- or semi-aquatic species such as birds and crabs. Tides also provide a mechanism for connectivity between wetland communities and the estuarine environment.
2. Intertidal areas (mudflats, mangroves, saltflats and saltmarshes) are only temporarily available to aquatic species. Therefore many species utilise resources as they become accessible. For example, many species such as juvenile barramundi *Lates calcarifer* are found in the highest densities on the uppermost fringe of the flood tide (J. Platten, QEPA, pers. comm., P. Stoneley, Sunfish, pers. comm.). Some species utilise the fringe of the flood tide for nourishment while they are high in the intertidal zone, for example, the fantail mullet *Valamugil georgii*.
3. Feeding activity might be less important for other species when they are in the intertidal zone, but presumably the presence of such species in the tidal fringe would indicate habitat utilisation rather than resource use, although this has not been demonstrated. An alternative suggestion is that such species may be incapable of swimming against the strong tidal currents when they are small and are swept into the intertidal zone involuntarily.
4. Tides play an essential role in the carbon transfer process. Habitats such as seagrass low in the intertidal zone have been shown as important in supporting food webs leading to fisheries production on the shore, for example, over mudflats in Port Curtis (Connolly & Guest 2004). Although the mechanism of carbon movement is still under study (e.g. Guest *et al.*, 2004), water movement is very likely to be important. For further details see Trophic links Diagram 7.4.
5. The continual flood and ebb of the tidal system flushes out areas such as mangroves and mudflats, removing excess salts, debris and detritus from the mudflats, mangroves, saltflats and periodically the saltmarshes (Wolanski *et al.* 2005). However, this has not been definitively demonstrated in Port Curtis.
6. Flood tides also provide for the oxygenation of soils in combination with bioturbation, as well as providing renewed food and nutrient resources to the area (Wolanski *et al.* 2005).
7. Tides are generally considered to maintain stability between the deposition and erosion of sediments, essentially providing a self-scouring mechanism that helps maintain the system in even balance (Wolanski *et al.* 2005).
8. Periodic inundation of wetlands provides connectivity between the marine environment and the brackish pools. Species of fish such as barramundi, bony bream and mullet are known to utilise this periodic connectivity to inhabit these brackish pools, where they may remain for long periods (P. Stoneley, Sunfish, pers. comm.). Marine connectivity is generally very unpredictable, but must occur under the correct conditions and at the right time for the connection to be effective, for example, during the period when juvenile barramundi are searching for suitable habitats (M. Sheaves, JCU, pers. comm.).
9. Sediments and other particulate matter originating from terrestrial sources are deposited in mangroves and other upper intertidal habitat via the tidal system rather than freshwater runoff (Furukawa *et al.* 1997).

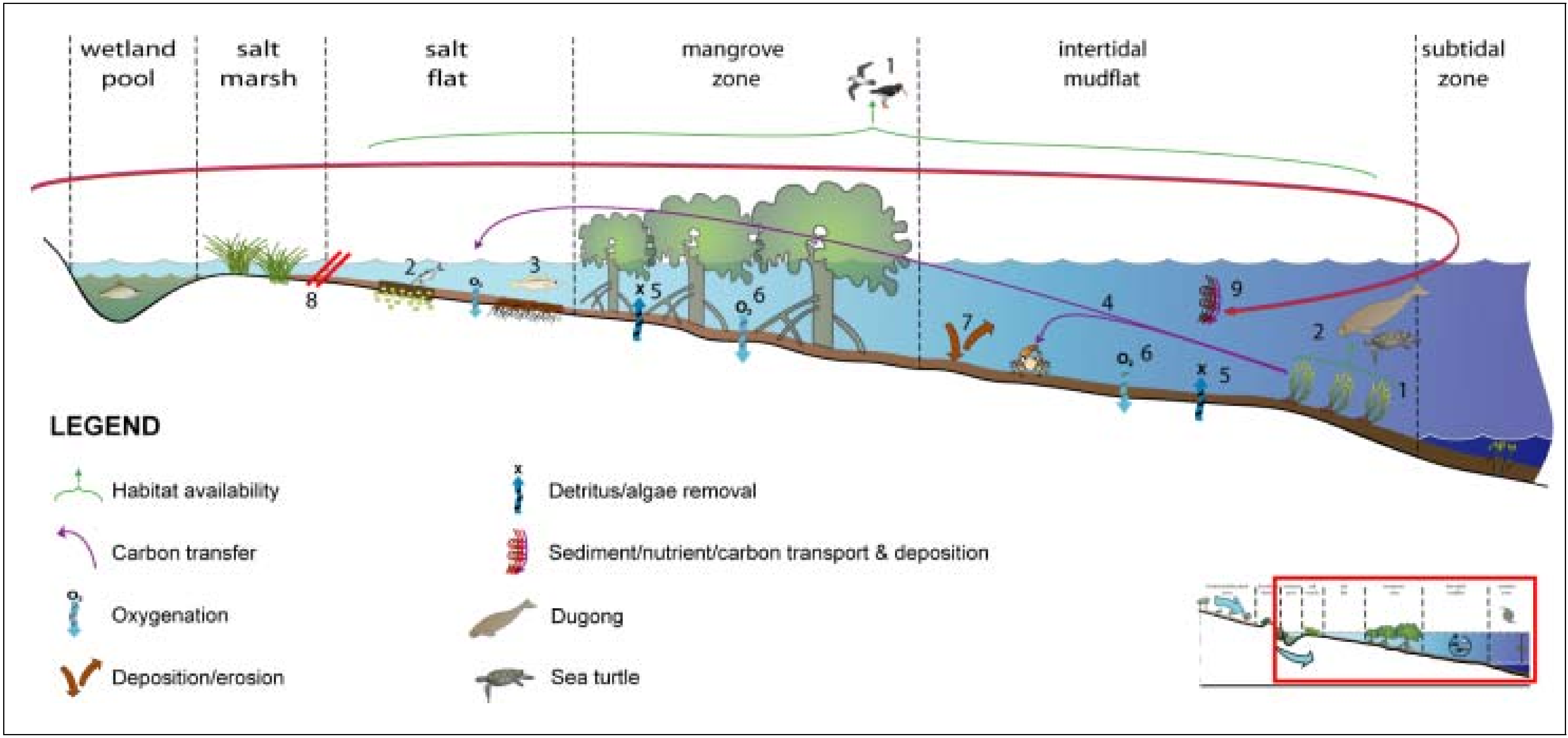

Diagram 7.2. Processes associated with tidal movement

Freshwater flow (Diagram 7.3)

Freshwater flow through intertidal wetlands has long been considered an important driving factor for many processes, including influences on recreationally and commercially important species in tropical Queensland such as banana prawns *Fenneropenaeus (Penaeus) merguiensis*, tiger prawns *P. esculentus*, mud crabs *Scylla serrata* and barramundi *Lates calcarifer*, among others (Robins *et al.* 2005).

Freshwater inflow to estuaries in tropical Australia generally increases during late spring (October, November) and summer (December–February), with less flow during the winter months (June–August). Port Curtis has two main freshwater inputs, namely the Calliope River and Auckland Creek, discharging directly into the semi-enclosed area of the harbour (Duke *et al*. 2003). The heavily impounded Boyne River, which discharges near the southern entrance to the harbour also contributes to the freshwater input into Port Curtis. There are also numerous other, smaller streams and creeks entering Port Curtis, including South Trees Inlet, Targinnie Creek, Graham Creek and Endfield Creek, all of which are ephemeral creeks, only providing freshwater input during periods of high rainfall.

An additional factor is the periodic incursion of fresh water from the Fitzroy River to the north of the harbour (Herzfeld *et al.* 2004). This occurs during periods of high rainfall, with water from the Fitzroy entering Port Curtis through The Narrows. Collectively, these freshwater inputs provide important processes for the intertidal environment of Port Curtis, although in some cases alteration of the system has led to some streams essentially being removed as potential sources of fresh water. For example, an intake for a power station causes salt water to flow upstream within Auckland Creek, with subsequent discharge into the Calliope River.

Overland freshwater flow (sheeting flow) is also deemed an important factor in the ecological functioning of the intertidal wetlands in tropical Queensland, although the nature of most of these relationships is not fully understood (Robins *et al*. 2005). Fresh water moving over terrestrial land not only helps replenish groundwater supplies through seepage, but also entrains sediments and nutrients, transporting them to the intertidal zone, where they are often deposited and utilised through the many processes occurring there.

1. Generally, groundwater inflow/outflow is important for the intertidal ecosystem. Studies within other intertidal ecosystems have shown that the process of bioturbation facilitates groundwater inflow, ventilating soils and bottom waters (Wolanski *et al.* 2005).
2. Groundwater outflow entrains algae and bacteria, removing them from the system (Wolanski *et al.* 2005). Bioturbation also flushes excess salts from the system, helping retain salinity at a habitable level (compare this process to drought conditions in model five).
3. The Calliope and Boyne Rivers provide habitat for important commercial species such as the banana prawn. This species is known to utilise the brackish zone of streams and rivers to some extent, particularly as postlarval nurseries (Ridd *et al*. 1988, Meager *et al*. 2003; Robins *et al*. 2005). After large rainfall and flood events, prawn densities within the brackish water nurseries are often reduced, suggesting a flushing or emigration into the greater estuary (Meager *et al*. 2003). This process is extremely important for replenishing stocks of commercially viable prawns (and other species, e.g. mud crabs) within Port Curtis (Robins *et al.* 2005).
4. Freshwater flow is known to influence the recruitment of species into estuarine systems, but the exact mechanism is unknown at this stage (Robins *et al.* 2005).
5. In years of very high flow into Port Curtis, invertebrate productivity is higher (Currie & Small 2005) and, subsequently, whiting catch rates are higher (Platten, see Chapter 4). Unusually, there is no lag. The higher catch rate is in the same year as the higher flow. The higher invertebrate productivity leads to faster growth in the youngest cohort so that they are caught as legal sized fish in their first year of life. Thus, in years of high flow, very large numbers of (on average) smaller fish are taken.

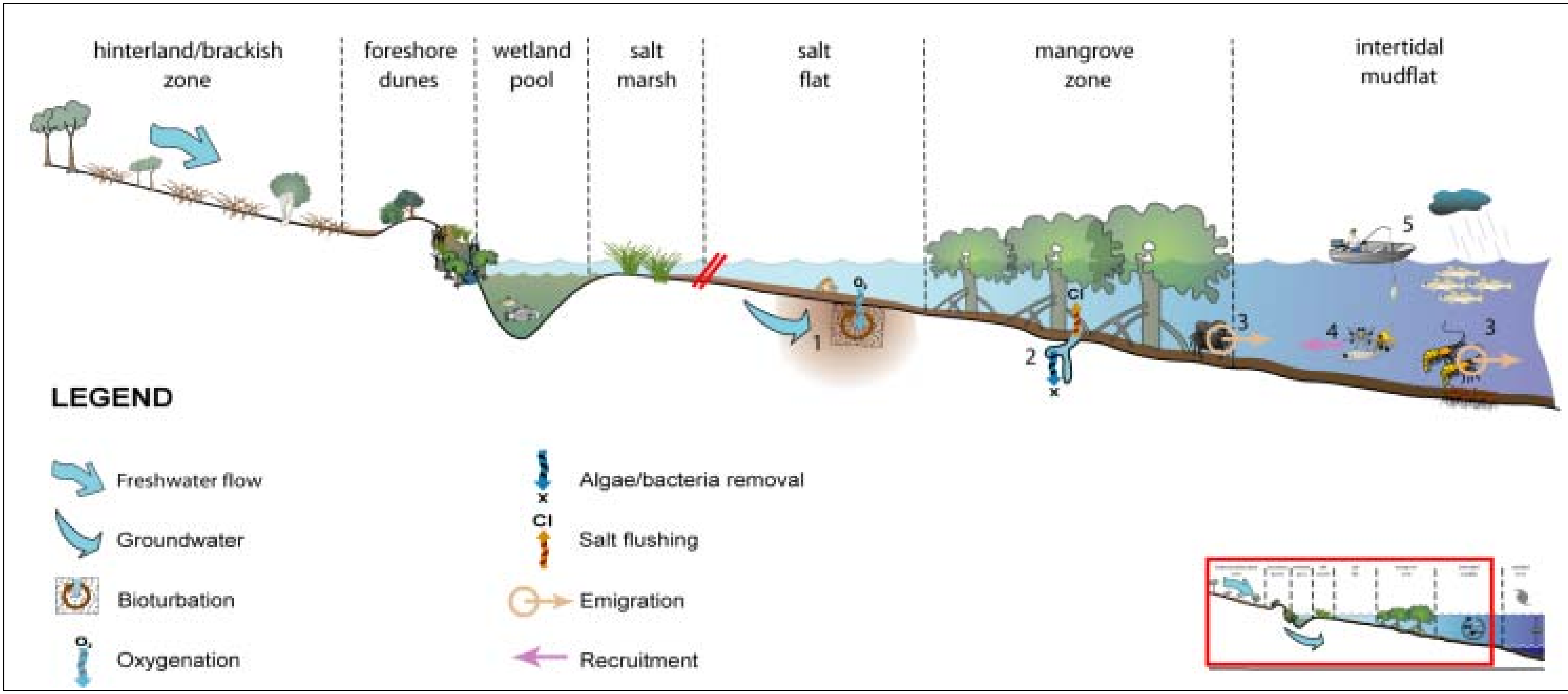

Diagram 7.3. Processes associated with freshwater flow

7.2.3 Trophic links (Diagram 7.4)

Trophic links are considered high on the priority list of issues for community stakeholders within the Port Curtis region. Fishing is one of the most accessible and participated in recreational activities in the Central Queensland region (Platten 2005). Gladstone, in particular, has one of the highest boat ownership rates per capita in Australia. Commercial fishing is also a major industry in Port Curtis, with up to 40% of Queensland's mud crabs coming from the Central Queensland coastline (Walker 1997).

Diagram 7.4 utilises some of the more pertinent species from Port Curtis to illustrate some of the trophic interactions that are known to occur within the intertidal zone there.

1. A study in Port Curtis showed that carbon utilised by higher trophic levels (e.g. fish, crabs and prawns) is largely derived from marine autotrophs, not terrestrial sources (Connolly & Guest in press).
2. For animals in much of the intertidal zone, including key fisheries species, carbon is supplied ultimately from seagrass meadows low in the intertidal or shallow subtidal zones. Connolly and Guest (2004) found that in Port Curtis the subtidal seagrass *Halophila* was more likely to provide carbon to intertidal crustaceans, while *Zostera* (intertidal) was a more important contributor to fish species. Mangrove carbon is not an important source for these animals (see Chapter 6).
3. Elsewhere it has been shown that mangrove carbon is important only for animals in the mangroves (Bouillon *et al*. 2004).
4. Upper wetland communities (e.g. brackish pools) are generally self-sustaining systems, producing and recycling carbon *in situ* (M. Sheaves, JCU, pers. comm.). Some carbon is also transferred via birds, periodic floods or tidal inundation, and the occasional movement of some animals over land.
5. There may be some role for microphytobenthos on mudflats in the Port Curtis food web either via direct grazing (e.g. *Valamugil georgi*) or a detrital loop. Further work is needed using novel chemical tracers to confirm the microphytobenthos role.
6. Port Curtis seagrass meadows are generally confined to the lower intertidal or shallow subtidal zones due to relatively high turbidity and low light penetration (Rasheed *et al*. 2002). Consequently, a mechanism or process exists by which carbon derived from seagrass is transported throughout the intertidal zone. This can either occur via indirect transport through trophic interactions or through direct transport where the seagrass is washed into the intertidal zone by the tides prior to being consumed. As yet, the nature and contribution of each of these transport processes are not well understood, although it is reported that most carbon fixed by seagrass enters the food web via the detrital system rather than through direct consumption (OzEstuaries Website, <www.ozestuaries.org>).
7. Seagrass is a major contributor to dugong and sea turtle diets (Rasheed *et al*. 2002). Port Curtis is formally recognised as providing dugong habitat and is a designated dugong protection area (Queensland *Nature Conservation Act 1992* and Queensland *Fisheries Act 1994).* However, the grazing potential within Port Curtis has not been quantified, although dugong and sea turtles frequent the area.
8. The rate of commercial and recreational harvest of certain key species, for example mud crabs, may result in cascading effects on lower trophic levels. This is an important theoretical consideration elsewhere but has not been addressed in Port Curtis.

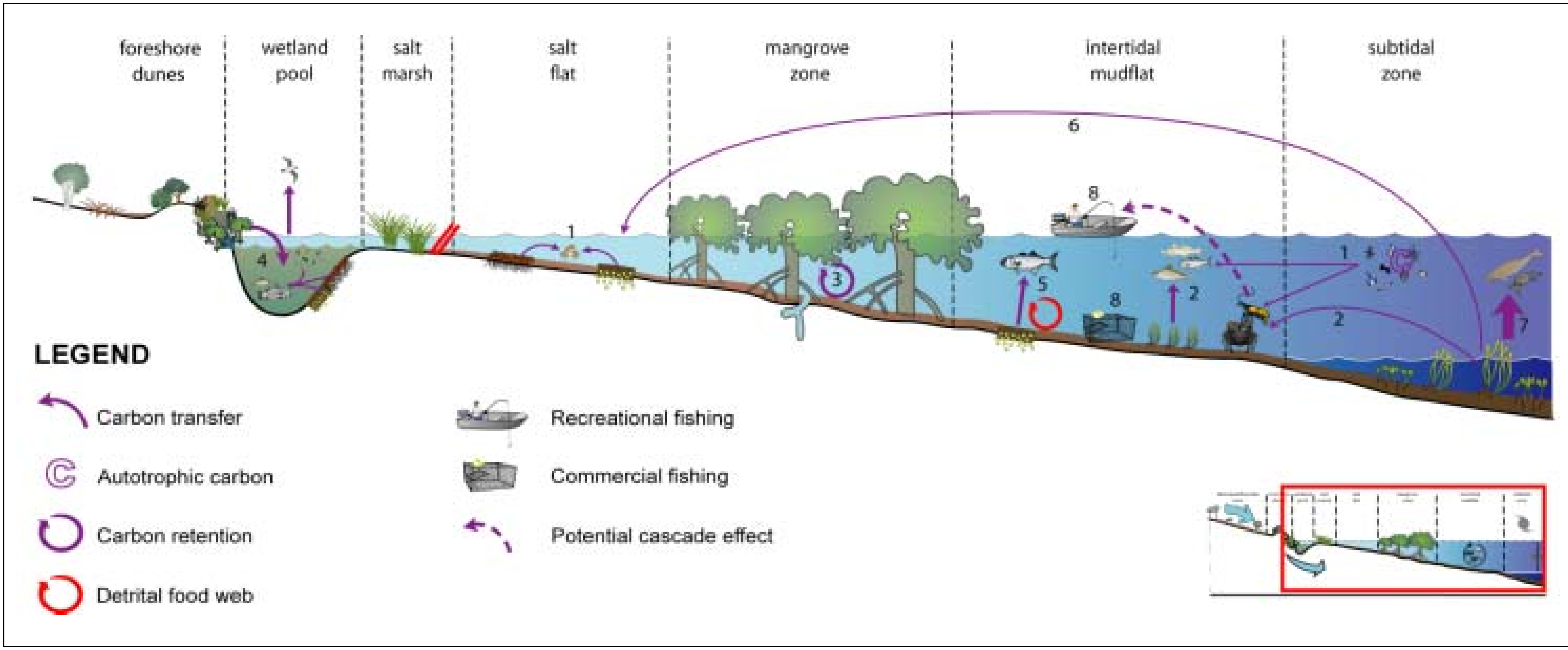

Diagram 7.4. Processes associated with trophic links

7.2.4 Storm and cyclone events (Diagram 7.5)

Severe storm events (e.g. cyclones) are relatively rare in the Port Curtis region, although it has been suggested that such events occur frequently enough to prevent species-stable communities (i.e. equilibrium) being reached (Saenger *et al*. 1988). Some of the more influential processes occurring in relation to storm events are illustrated in Diagram 7.5. It should be noted that very little information is currently available on the impacts that major storm events have on the Port Curtis intertidal ecosystem as a whole.

1. Severe storm events may induce scouring, particularly in rivers, streams and intertidal channels. Such scouring events are considered important for resetting the system (Saenger *et al*. 1988). However, in some instances scouring may also remove or impact on valuable resources such as seagrass.
2. Scouring of rivers, streams and channels causes drastic reductions in macrobenthic communities. Succession does not necessarily follow such events, but is more likely a process of recolonisation by organisms that were there prior to the storm/cyclone event (Saenger *et al*. 1988). Initial recolonisation may occur relatively rapidly (in the order of months) while reaching a stable community or a quasi-steady state can take in excess of 10 years (Saenger *et al*. 1988).
3. Nutrient and sediment runoff as a result of a storm event is increased markedly, thereby elevating nutrient levels in the water column (Currie & Small 2005).
4. The elevated nutrient levels may result in plankton blooms, although factors such as light availability and forced water-column mixing may hinder any such blooms.
5. High levels of sediment in runoff water, as well as scouring/erosion from high wind and wave action, increases turbidity in the water column (Currie & Small 2005). This may increase sediment deposition and may result in seagrass dieback (see Trophic links information for the importance of seagrass). This may occur due to smothering from sedimentation processes, or may be a result of reduced light penetration due to increased turbidity (Rasheed *et al*. 2002).
6. Fine sediments combine together with the mucus of plankton, creating flocs, which may settle out of the water column more rapidly. This process also removes some of the nutrients that are attached to the sediments.
7. Storm-induced dieback of mangroves has been reported in some areas within Port Curtis, for example, with *Ceriops* and *Rhizophora* species showing considerable mortality after a severe hail storm in 1994 (Houston 1999). Duke *et al*. (2003) suggests that mangroves may be susceptible to storm damage, with dieback after hail storms, cyclones, flooding and strong winds being reported in various different locations worldwide. However, little data is available for the effects of cyclones on mangrove survival in Port Curtis.
8. During severe storm events, groundwater supplies are replenished over a very short period, and wetland pools are likely to be pushed more towards fresh than saline conditions, depending on the physical size and location of the pool (see Freshwater flow Diagram 7.3 for groundwater effects).

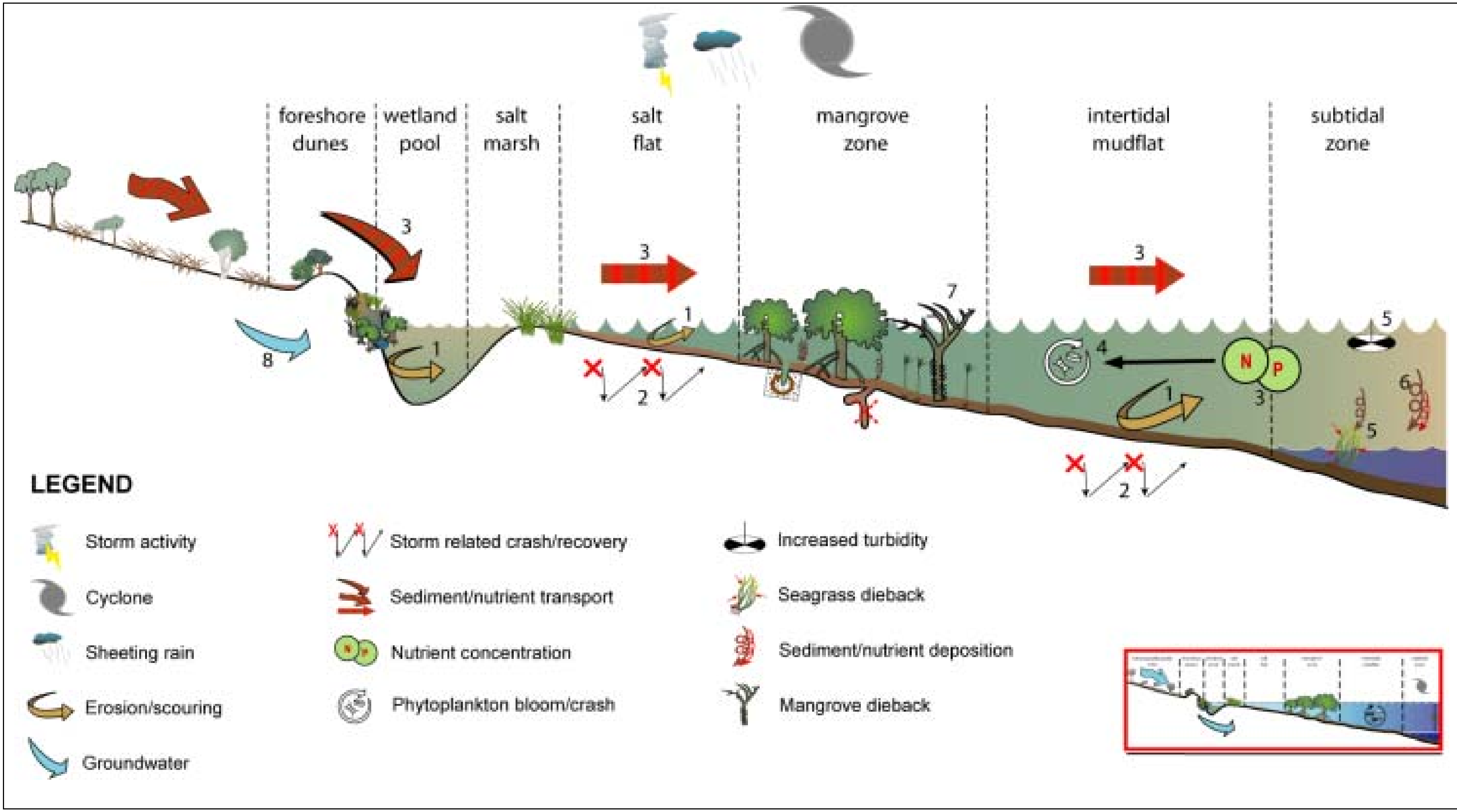

Diagram 7.5. Processes associated with storm and cyclone events

7.2.5 Drought conditions (Diagram 7.6)

Port Curtis experiences low but highly variable rainfall. Occasionally, below-average rainfall can lead to drought conditions, with current weather patterns indicating a trend to more frequent droughts in the future.

1. Desiccation stress in intertidal communities increases due to prolonged exposure to dry conditions over the summer months. Under normal conditions this period is the wet season, where pore water and intertidal flats (particularly saltflats) are more regularly inundated with fresh water.
2. Salinity in saltflat sediments has been found to be up to five times higher than in ambient sea water (Teasdale *et al.* 2005, Chapter 3). Without periodic freshwater input, salinity may increase further due to evaporative forcing (Currie & Small 2006). Elevated salinities may render the area uninhabitable, even for the hardiest of species (Teasdale *et al*. 2005, Chapter 3).
3. Ecotone shifts in intertidal wetlands may occur after long-term changes to climate, such as droughts (Duke *et al*. 2003). When fresh water is scarce, mangroves on the fringe of the saltflats may die back, increasing the area of saltflat in comparison to area of mangroves (Duke *et al*. 2003). This trend may reverse over extended periods of high rainfall. The time frame over which such ecotone shifts may occur has not been quantified, although it would be expected to be over the very long term (decades) rather than short-to-medium time frames.
4. During drought conditions the cue for species (e.g. barramundi and prawns) to emigrate or breed may be lost or reduced, resulting in less recruitment to the nurseries (Robins *et al*. 2005).
5. Mud crabs and prawns may remain sheltered in mangroves or brackish water environments when there is a lack of fresh water (Robins *et al*. 2005).
6. The absence of freshwater cues may have important implications for commercial and recreational fisheries (e.g. prawns), albeit with a lag period. If nurseries and/or recruitment processes are adversely affected by drought conditions, the biomass available in fishing areas may decrease over subsequent seasons (Robins *et al*. 2005). This phenomenon can be compared to that where higher freshwater flows result in higher fish catches in the same year (see Diagram 7.3).
7. Water depth in brackish wetland pools will decrease and salinity increase due to evaporation. This can affect the system through:
   - habitat loss – as the water level decreases less habitat is available for shelter and food. Individuals living in the pool are therefore more easily captured by predators, including wading birds (M. Sheaves, pers. comm.).
   - pool salinity may reach levels that are uninhabitable by some species, although in larger pools this is unlikely.
8. Saltwater incursion into brackish zones and groundwater may affect some of the less salt-tolerant species. Stephenson and Cook (1979) found that the duration of salinity changes was more important than the magnitude. As such, it is possible that many species may tolerate seasonal incursions of salt water (e.g. during the dry season), but may become susceptible if the incursion lasts for extended periods, such as during a drought.
9. Evapotranspiration by flora, particularly mangroves, may reduce groundwater supplies and consequently the ability to remove bacteria and algae from the system (see Diagram 7.3). This may also have the effect of increasing acidity and anoxia in soils (Wolanski *et al*. 2005).
10. Nutrient input from terrestrial sources will decrease, forcing aquatic primary production to become almost wholly supported by nutrients already available in the system. Terrestrial sources of sediment and organic matter, whether discharged from rivers and deposited from tidal current or sourced directly from the terrestrial fringe of the mangroves, will be reduced (Furukawa *et al*. 1997)
11. Erosion/resuspension of sediments in upper intertidal and sheltered areas may decrease or cease, depending on tidal forcing and wave motion.

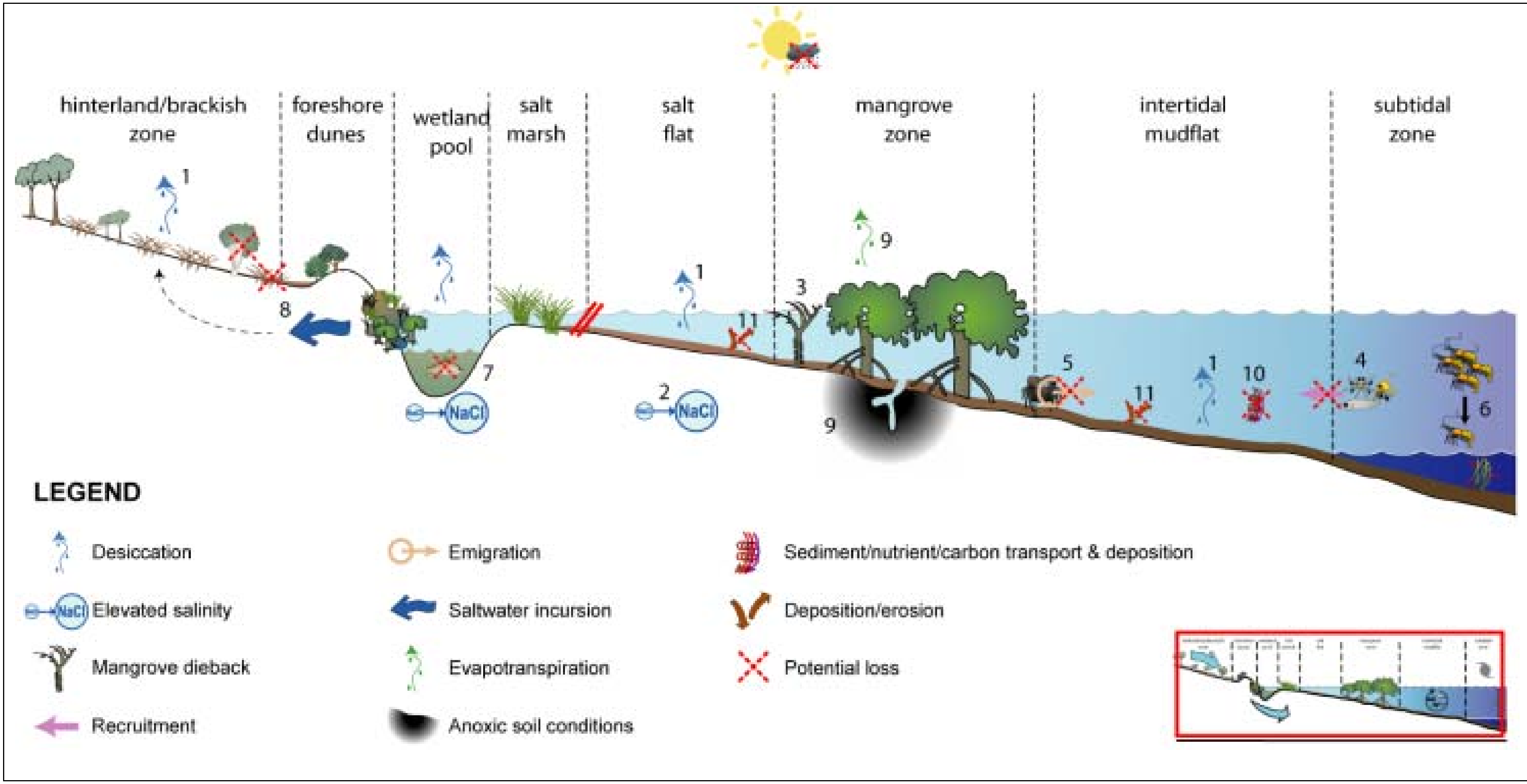

Diagram 7.6. Processes associated with drought conditions

7.3 Pressures impacting on the system

Apart from the processes acting on and driving the system, numerous pressures also operate within the intertidal wetlands. These pressures are either acting on the processes themselves, for example dams and weirs, or acting on the overall system, for example contamination. Again, the key pressures have been identified and conceptual models developed to illustrate the impacts of such pressures. The key pressures have been identified as the following:

1. **Land infill** – Physical creation of new land over the saltflats, mangroves and intertidal areas to facilitate further construction or development. This process generally involves the creation of bund walls with sediments and infill from varying locations and sources placed behind the bund wall to create the new land. This often results in diminishing the extent and changing the nature of habitats.
2. **Dredging** – Although Port Curtis is a naturally deep harbour, dredging activities are a necessary requirement for the safe passage of shipping and for the strategic growth of the Port facilities. Dredging may influence sediment dynamics (turbidity and particle size composition) within the estuary through the removal of large areas of sediment, and may change current flow through the creation of channels.
3. **Barriers, dams and weirs** – The construction of physical structures to contain large quantities of fresh water for use in community and industrial applications, as well as the construction of causeways for roading and railways, and other infrastructure. This is particularly relevant in the dry tropics where fresh water is an increasingly scarce commodity, while human populations continue to increase.
4. **Contaminants** – The large number of industrial and municipal activities occurring within Port Curtis increase the potential for contaminants or novel chemicals to be released into the local ecosystem. Often such pollutants have the potential to remain in the system for long periods, and some may accumulate in certain situations. It should be noted that airborne contaminants also have the potential to enter the intertidal ecosystem.
5. **Climate change** – An alteration in the climate has far-reaching implications for all regions, ecosystems and activities, including of course the Port Curtis area. Although exact processes and pressures are not known, a conceptual model has been developed to illustrate the potential effects on the intertidal wetlands of Port Curtis.

7.3.1 Pressures overview diagram

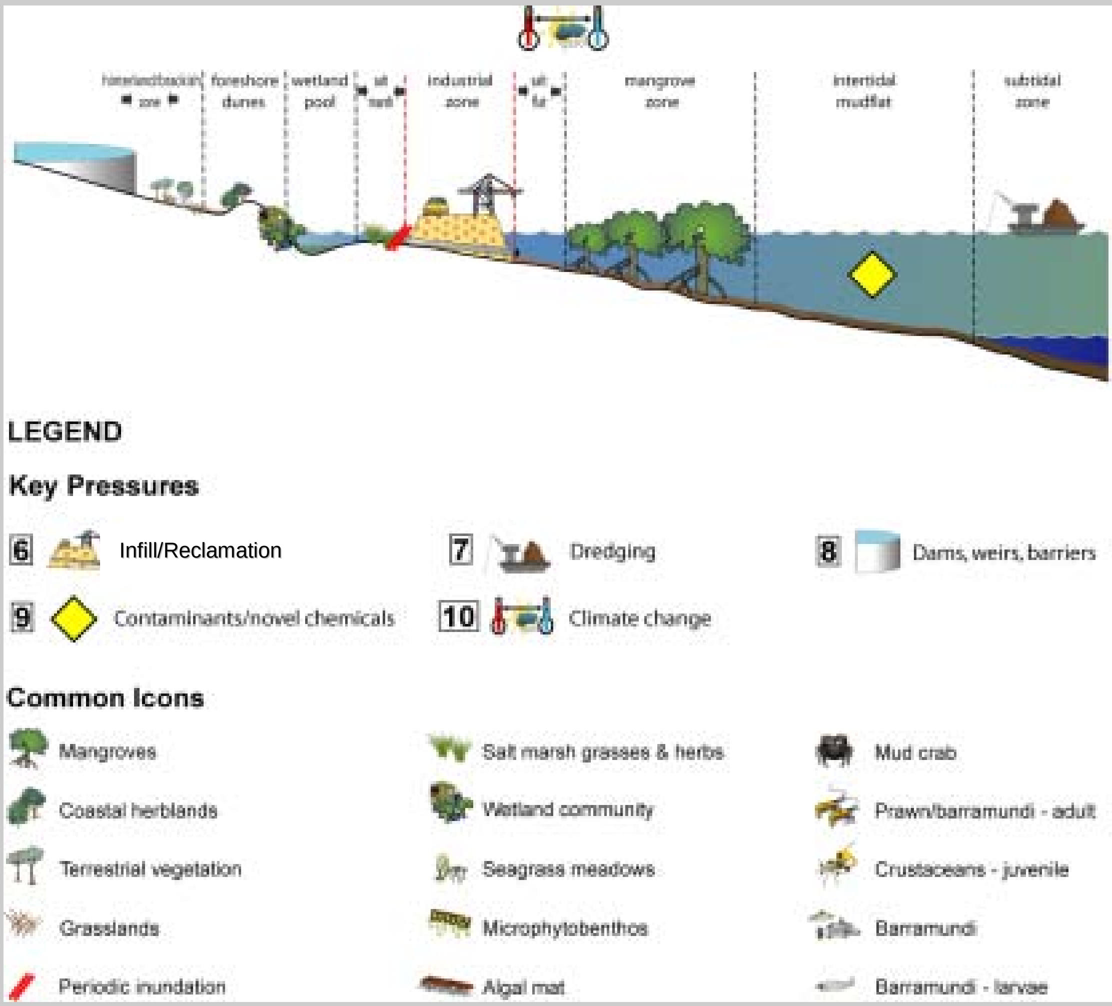

Diagram 7.7. Overview of key pressures on intertidal wetlands

The above diagram represents an overview of the key pressures identified as most important for the Port Curtis intertidal wetlands: infill; dredging; dams, weirs and barriers; contaminants; and climate change.

Each of the following conceptual diagrams includes a key to the icons particular to that diagram; however the legend in the overview model above describes the main icons, such as mangroves, saltmarsh grasses, and mud crabs, that are common to all the process diagrams and which are not shown in each individual icon key.

Each diagram is also accompanied by numbered text to describe the pressure depicted in that model. In addition, the width of the arrows on each diagram indicates the relative magnitude of a pressure.

7.3.2 Infill (Diagram 7.8)

The ongoing development of port facilities within Port Curtis requires additional land. In order to achieve this, the main process to date has been to construct bund walls and infill using material from varying locations, for instance, dredged material from the harbour or terrestrial material from other land-based developments. In some early instances of infill, industrial waste products and even old refuse tips have been used as bases for land infill.

Infill was identified by stakeholders as being the main pressure of concern in the Port Curtis intertidal area.

The diagram illustrates the known and suspected impacts and influences of infill on intertidal wetlands. Although this model deals just with intertidal wetlands, it is important to recognise that such impacts and influences are not confined to the intertidal zone, but have implications for all areas within Port Curtis.

1. Mangrove communities are often removed as a consequence of infill and the development of infrastructure corridors. A 38% reduction in mangrove area between 1941 and 1999 is largely attributed to infill activities (Duke *et al*. 2003).
2. The saltmarsh may be completely removed from the system in some instances of infill, but is reduced in size, and presumably function, in other instances. Historically, the loss of saltmarsh from infill appears to be higher than for mangroves (Duke *et al*. 2003).
3. Materials used for construction of bund walls and the subsequent backfill may contain contaminants not generally found in the intertidal zone (see Contaminants Diagram 7.11 for implications). This may include naturally occurring chemicals/substances of terrestrial origin. When leached into the marine environment in concentrated forms these substances may become contaminants (i.e. novel chemicals), for example iron compounds. Acid sulfide exposure from construction activities and subsequent leaching into the marine environment is also a potential threat. To date, adverse consequences on the macrobenthic communities (and presumably the microphytobenthos) from land infill have been difficult to determine due to th paucity of physicochemical data (Currie *et al*. 2003). Studies by Central Queensland University are continuing into the effects of landfill contaminant in Port Curtis.
4. Altered hydrology (e.g. the creation of highways and railways across saltflat reduces the penetration of salt water into the saltflat area, effectively isolati such areas from tidal inundation. Although pipes are included to allow the transfer of water, the net effect on the ecology of the saltflat is not known. F instance, retention time of water in the saltflat may be increased due to restriction of flow pathways by pipes. This may in turn negatively affect the saltflat communities through prolonged exposure to high salinity water as opposed to periodic inundation.
5. Fish migration via natural channels is no longer possible due to land infill.
6. Groundwater penetration is reduced due to the collection of urban runoff in stormwater systems.
7. The increased area of developed land leads to increased agricultural and urban runoff. Urban runoff is also concentrated into point-source outfalls rather than diffuse sources as would occur naturally. Sheeting flow of fresh water over the saltflats and mudflats and filtration through mangrove system is reduced.
8. The effect of infill on the physical characteristics of the intertidal zone is also unclear, although Teasdale *et al*. (2005) in Chapter 3 did not find any significant differences between sites close to (Fisherman's Landing) or far from (Kangaroo Island) infill activities.

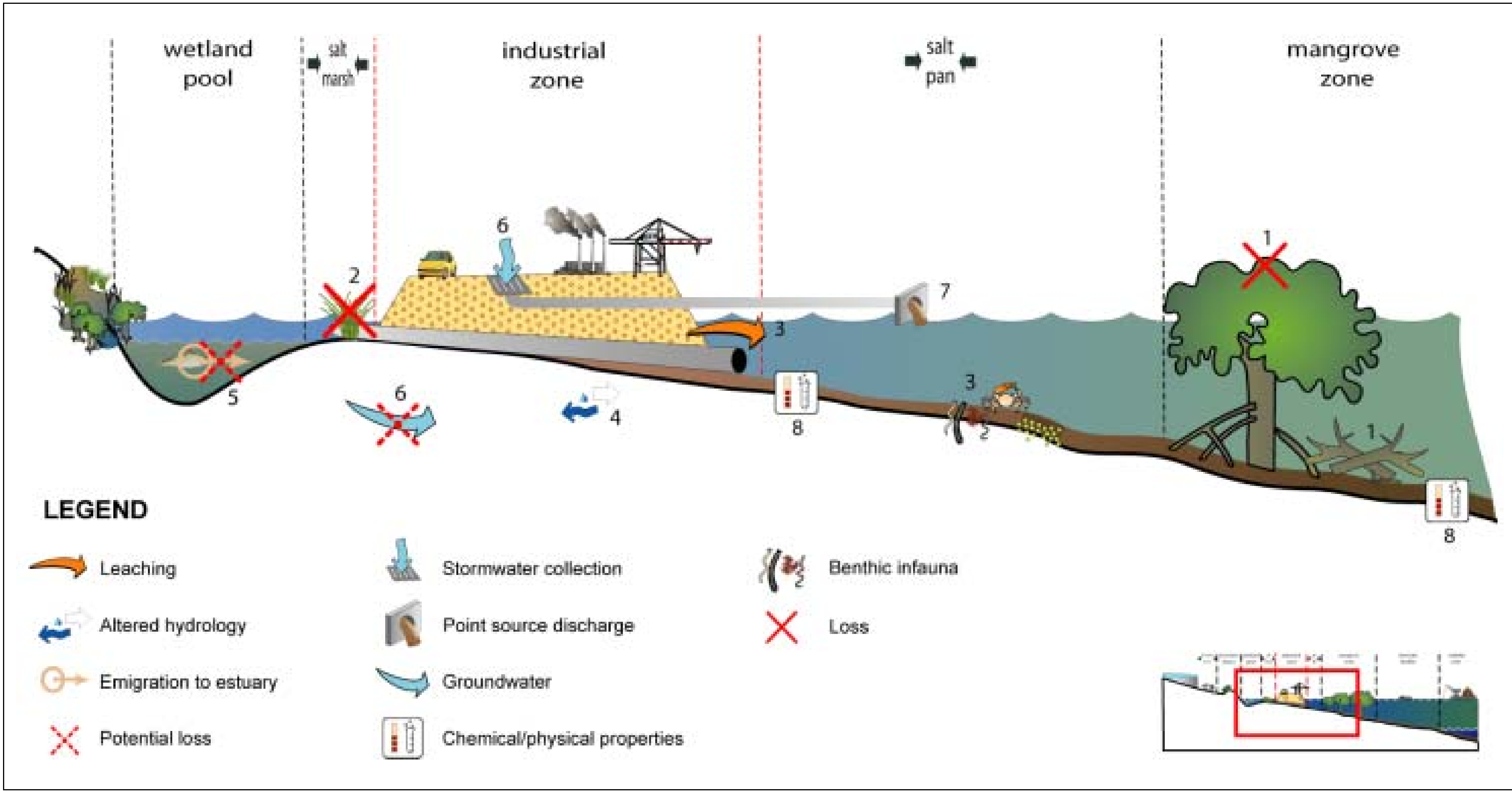

Diagram 7.8 Pressures associated with the infill of intertidal areas

7.3.3 Dredging (Diagram 7.9)

Dredging of the Port Curtis harbour is a necessary aspect of maintaining the Port in operational condition (Gladstone Port Authority 1997). At this stage dredging occurs periodically, with perhaps 18 months to two years gap between maintenance dredging. When new facilities are developed, major dredging is required to ensure safe channels are provided for the new infrastructure. Most of the environmental concern is focused on major dredging activities, although both types of dredging have potential impacts on the intertidal zone. Most dredging operations occur in the subtidal environment, but have the potential to impact on the intertidal areas within Port Curtis.

1. The process of dredging physically removes the upper layers of sediment, benthic infauna, microphytobenthos and seagrass communities, leaving less inhabitable sediments such as anoxic layers. The effect of this removal on the system is not well quantified, although a study by Saenger *et al*. (1988) in Port Curtis suggests that where benthic communities are removed, it may take months to see any recovery and many years to reach quasi-stable communities.
2. Turbidity may increase due to the resuspension of fine sediments during the dredging process, leading to effects being experienced in areas adjacent to the dredging zone. However, the level of change to turbidity and sediment settlement rates as a direct result of dredging is difficult to determine due to the naturally relatively turbid waters of Port Curtis. Monitoring to assess the effects of dredging is continuing by Central Queensland University.
3. If turbidity is increased, light penetration will decrease, thus lowering seagrass productivity (Rasheed *et al*. 2002). This is likely to have important trophic ramifications for fish in Port Curtis (see Diagram 7.4).
4. The life cycle of many important recreational and commercial species may be adversely affected by dredging. For example, seagrass is an important habitat for prawn species (Robins *et al.* 2005), and may also harbour juvenile crabs and fish. It is also an important carbon source for mud crabs and many fish species (see Diagram 7.4 for more detail).
5. Large iconic herbivores such as dugong and sea turtles may also be affected by the loss of seagrass due to dredging. For example, Preen and Marsh (1995) found that dugong numbers in Hervey Bay plummeted (~1753 ± 388 reduced to 71 ± 40) after floods and a cyclone removed large areas of seagrass meadows.
6. Saenger *et al*. (1988) also note that trophic groups change as the ecosystem matures. This observation may have important implications for fishery species in Port Curtis, particularly if the correct trophic groups are not able to establish themselves due to repeated dredging activities.
7. It is suggested that a potential increase in the level of contamination may be observed after dredging operations due to many years' worth of sediment-linked contaminants being resuspended over a very short time frame. Van Den Berg *et al*. (2001) found that trace metal concentrations increased in the suspended matter, whereas water column levels were not significantly affected by contamination. This is partly due to the rapid chemical transformation of potential contaminants such as sulfides (Vale *et al*. 1998). However, nutrient levels were found to be significantly higher in the vicinity of dredge equipment (Lohrer & Wetz 2003). It was also noted that the level of contamination as a result of dredging depends on the sediment contaminant levels.
8. Depending on dredging frequency, an equilibrium in community structure may not be reached (Saenger *et al*. 1988). Consequently, populations could be unstable and more susceptible to both natural and anthropogenically induced disturbance.

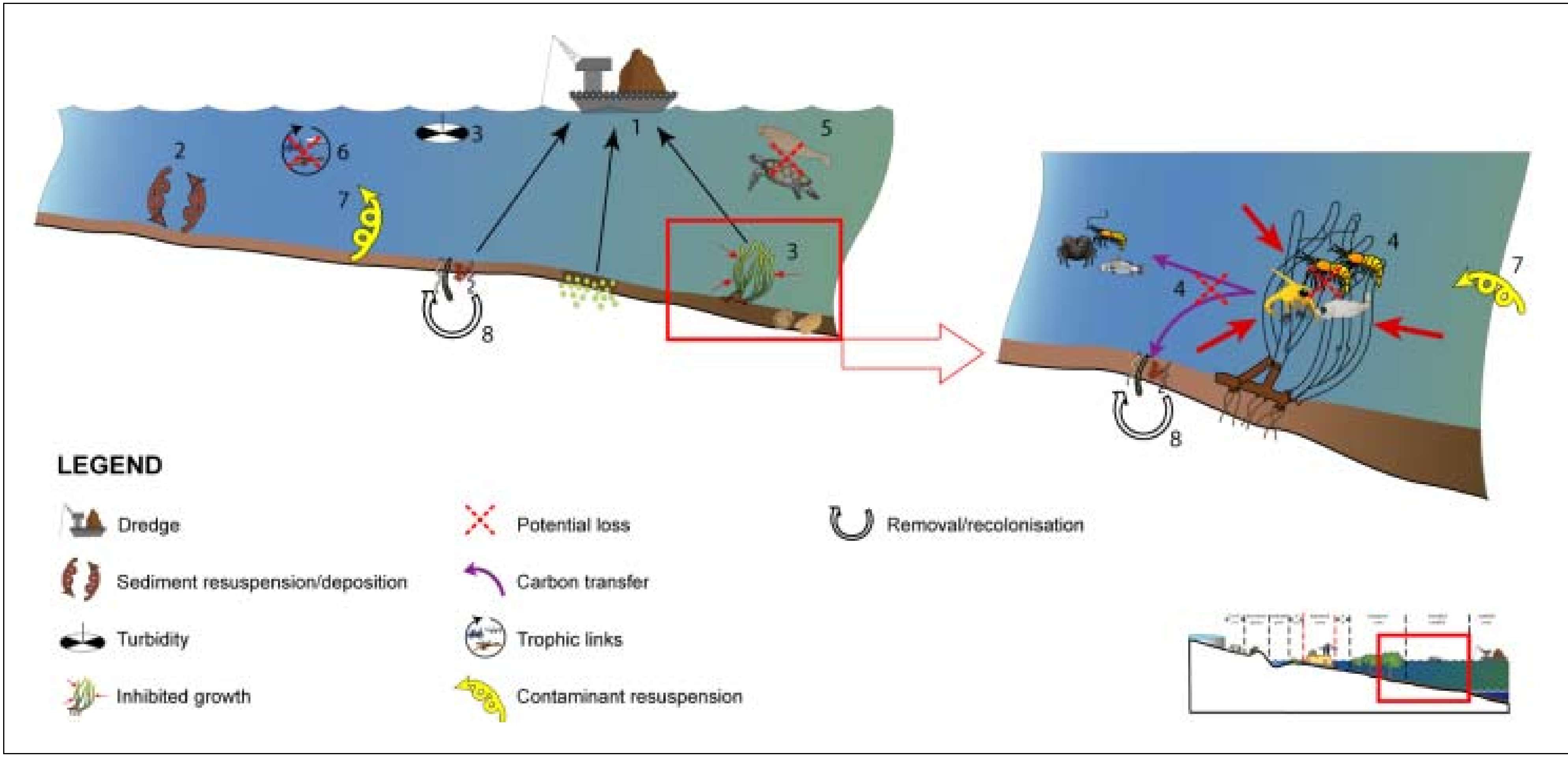

Diagram 7.9. Pressures associated with dredging

7.3.4 Dams, barriers and weirs (Diagram 7.10)

Within the Port Curtis region, major efforts are put into trapping water supplies while they are available and storing that water for later use. To date, the most used method is the construction of dams and weirs on the major streams and rivers, for example, the Boyne River. The construction of roads, railways and infrastructure corridors can also have a barrier effect. While these structures provide for a more secure source of fresh water and provide essential infrastructure, the implications for downstream communities are far-reaching, including for the intertidal zone. At this stage, the Calliope River, the main freshwater input into Port Curtis, has little in the way of barriers, dams or weirs restricting the natural flow.

1. Downstream freshwater flows are reduced in comparison with natural conditions, leading to a reduction in the area of the brackish zone. For example, prior to damming, the Boyne River had a brackish zone in the order of several kilometres long. This zone has now been reduced to the order of hundreds of metres long (J. Platten, QEPA, pers. comm.).

2. Groundwater flow may be reduced, not only in the vicinity of the barrier, but downstream as well. Lack of freshwater input to the groundwater system may eventually lead to a saltwater incursion, which can have a detrimental effect on the less salt-tolerant terrestrial species such as the *Melaleuca* species.

3. Fresh water entering the intertidal system is reduced (see Drought Diagram 7.6 for associated implications).

4. The flow of nutrients and sediments usually available to the intertidal zone from river flow in Port Curtis may be reduced (Herzfeld *et al*. 2004).

5. Studies have shown that freshwater flow influences estuarine communities in a positive way (Currie & Small 2005; Robins *et al*. 2005). For example, a drought in 1997–1998 induced a reduction in invertebrate numbers and diversity within Port Curtis by at least half (Currie & Small 2005). Consequently, the extraction or diversion of fresh water from within the catchment may have far-reaching implications for estuarine communities.

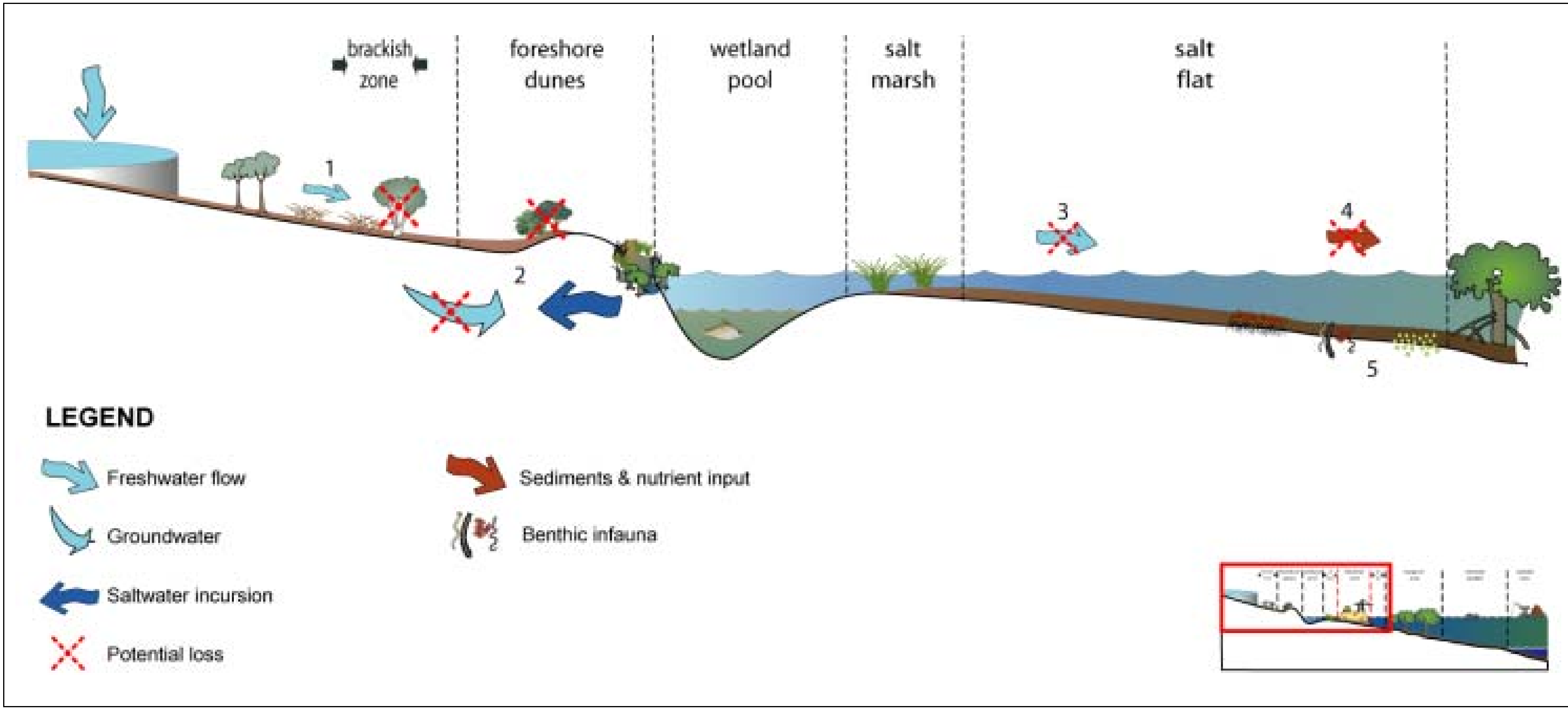

Diagram 7.10. Pressures associated with dams, barriers and weirs (includes roads, railways and infrastructure corridors)

7.3.5 Contaminants (Diagram 7.11)

Contamination of Port Curtis with pollutants and novel chemicals is a concern, both to industry, stakeholders and the general community. Larger industries in the Port Curtis region generally take reasonable steps to avoid contamination of the local environment, although 100% avoidance is not possible. However, Port Curtis is relatively clean for such an industrialised area and this is directly attributable to the efforts of the major industries of the area. Much of Gladstone's wastewater is now recycled, although its release into the Calliope River has been problematic in the past. Nevertheless, some contaminants manage to find their way into the system, eventually ending up in the marine environment.

1. High levels of fine dust and industrial discharge into the atmosphere have the potential to cause acid rain, thereby impacting on much greater areas than just the intertidal zone.
2. Land use practices (urban, industrial and agricultural) within the catchment area have the potential to impact on the intertidal zone. Land clearing generally increases terrestrial runoff, which may also contain pesticides, herbicides and other novel chemicals. These chemicals find their way into the groundwater or through the intertidal wetlands and eventually to the estuary-based intertidal zone, where they may impact on various species.
3. Contamination levels may be periodically elevated due to urban runoff during periods of rainfall. Although not visible, residue on city streets, roads and even rooftops is likely to wash out into the estuary via the stormwater system, providing point sources of contamination.
4. Purported leaching from land infill in Gladstone may also contribute contaminants to the water column (see Section 7.3.2, point 1). The long-term effects of leached contaminants within Port Curtis are currently being investigated by Central Queensland University.
5. The uptake of certain contaminants by mangroves has been well documented, particularly effluent-related contamination (Jones *et al*. 2001; Costanzo *et al*. 2004).
6. Filter-feeding organisms such as oysters, mussels and clams are known to accumulate contaminants and toxins in their tissues (Andersen *et al*. 2003a,b). Levels of contamination in higher trophic species have not been extensively researched, although it has been documented that the carnivorous mud crab *Scylla serrata* had higher levels of heavy metals than did lower trophic level fiddler crabs *Uca coarctata* (Andersen *et al*. 2004).
7. Seagrass is known to accumulate contaminants, potentially to levels where it may become sublethal (i.e. seagrass loss is likely to occur) (Prange & Dennison 2000). Contaminant accumulation in seagrass stocks may have ramifications for higher trophic assemblages, as the contamination is likely to accumulate exponentially in higher trophic level organisms.
8. Tributyltin contamination (TBT, linked with shipping activities) was found to adversely affect the mulberry whelk *Morula marginalba*, most notably affecting the reproductive organs of females (Andersen 2004). However, results within Port Curtis were not high relative to other ports that have been studied. Between 1987 and 1990, all States and Territories in Australia either restricted the use of TBT antifouling paint on vessels over 25 m length, or reduced the permissible leaching rate of TBT. Sales have also been restricted (EPA 2003).
9. There is also potential for pollutants to enter the intertidal system directly from the atmosphere. However, this has not been quantified within Port Curtis.

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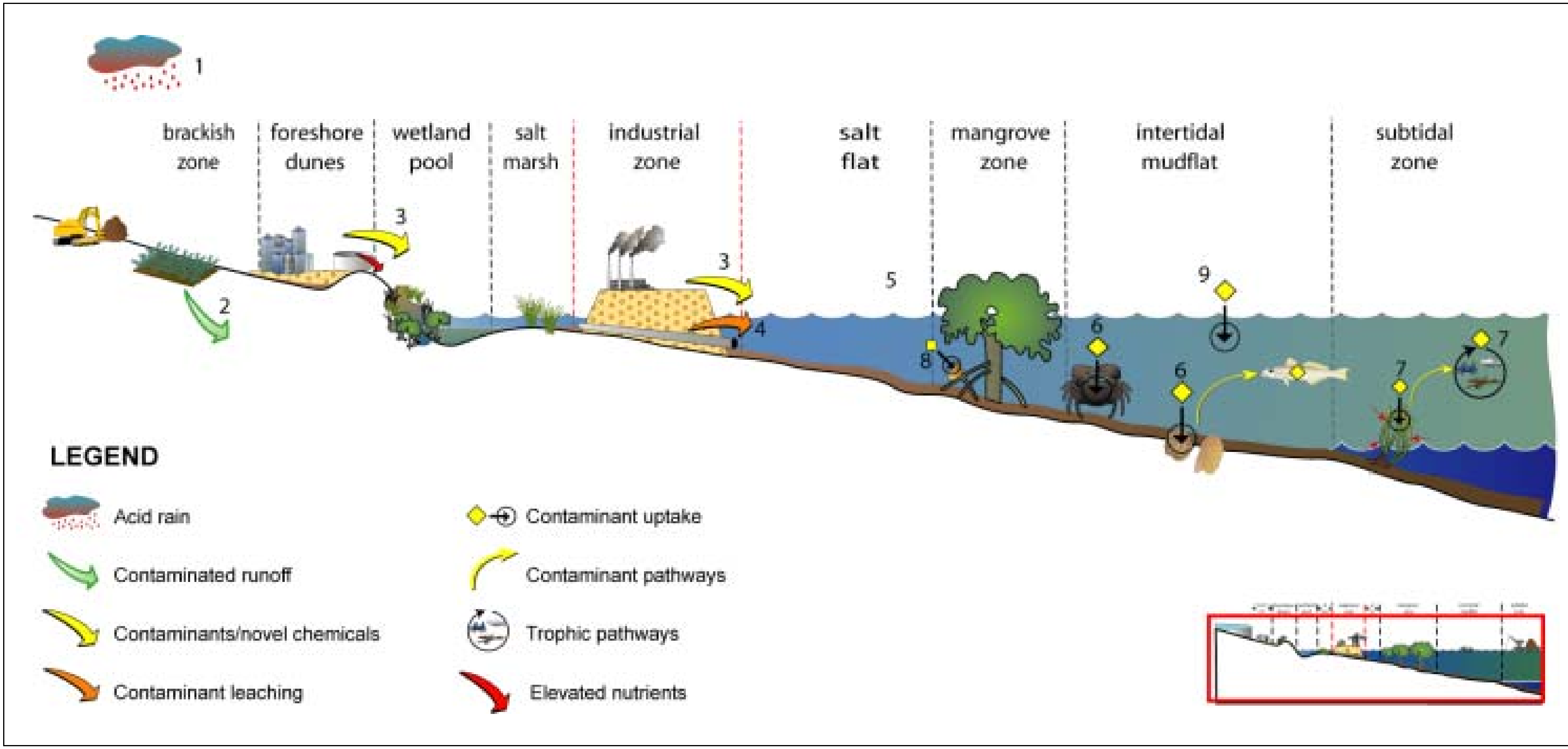

Diagram 7.11. Pressures associated with contaminants entering the system

7.3.6 Climate change (Diagram 7.12)

Climate change is a global phenomenon that has many implications worldwide. Although it is not known exactly how many systems would be affected during such changes, a number of inferences can be made from the information we have. Some of the following instances are potential outcomes from accelerated climate change. The model shows impacts in a very exaggerated way, but the severity of impacts can at this stage only be guessed.

1. Elevated temperature is the most prominent climate change characteristic, as it imparts an influence on most other aspects of the climate and associated effects.
2. Within Port Curtis a greater tidal surge could be expected as result of increased temperature and rising sea level.
3. As a result of the rising sea level, loss of mangrove zones will become pronounced. The current mudflats, mangrove and saltflat zones would eventually become subtidal. Saltmarshes and wetland communities would disappear. However, depending on the rate of change, the mangroves may be able to progressively establish elsewhere.
4. The effect of this change on aquatic organisms is unknown, although with the increased temperatures it would be expected that the ecosystem would tend more towards a tropical assemblage than subtropical, as it is at present. Again, this process may depend on the rate of change, as some species may be able to adapt to the altered conditions.
5. A more rapid turnover of seagrass meadows is expected due to the higher tidal surge and increased frequency of storm activity.
6. Increased tidal surge may increase productivity, or may inhibit seagrass growth altogether. As long as food is available, climate change should have minimal impact on dugong and sea turtles as these species inhabit much warmer climes at present.
7. Saltflats may form on the tidal fringes. However, the topography of the existing terrestrial landscape in Gladstone may prevent large areas of saltflat forming. Instead the mangrove-dominated system of today may tend more towards a sandy, pebble or rocky dominated shoreline at the base of the foothills.
8. Large areas of land infill may become inundated, eventually changing to intertidal areas. However, construction to increase the height of infill may be able to compensate for the rise in sea level. Protective barriers may need to be established to help mitigate the effects of severe storms.
9. The entire system would be expected to experience more frequent, high-intensity storms (e.g. cyclones), followed by extended dry periods. As such, the seasonal influence and spread of freshwater input would be removed, with freshwater input occurring in large quantities over short time frames.

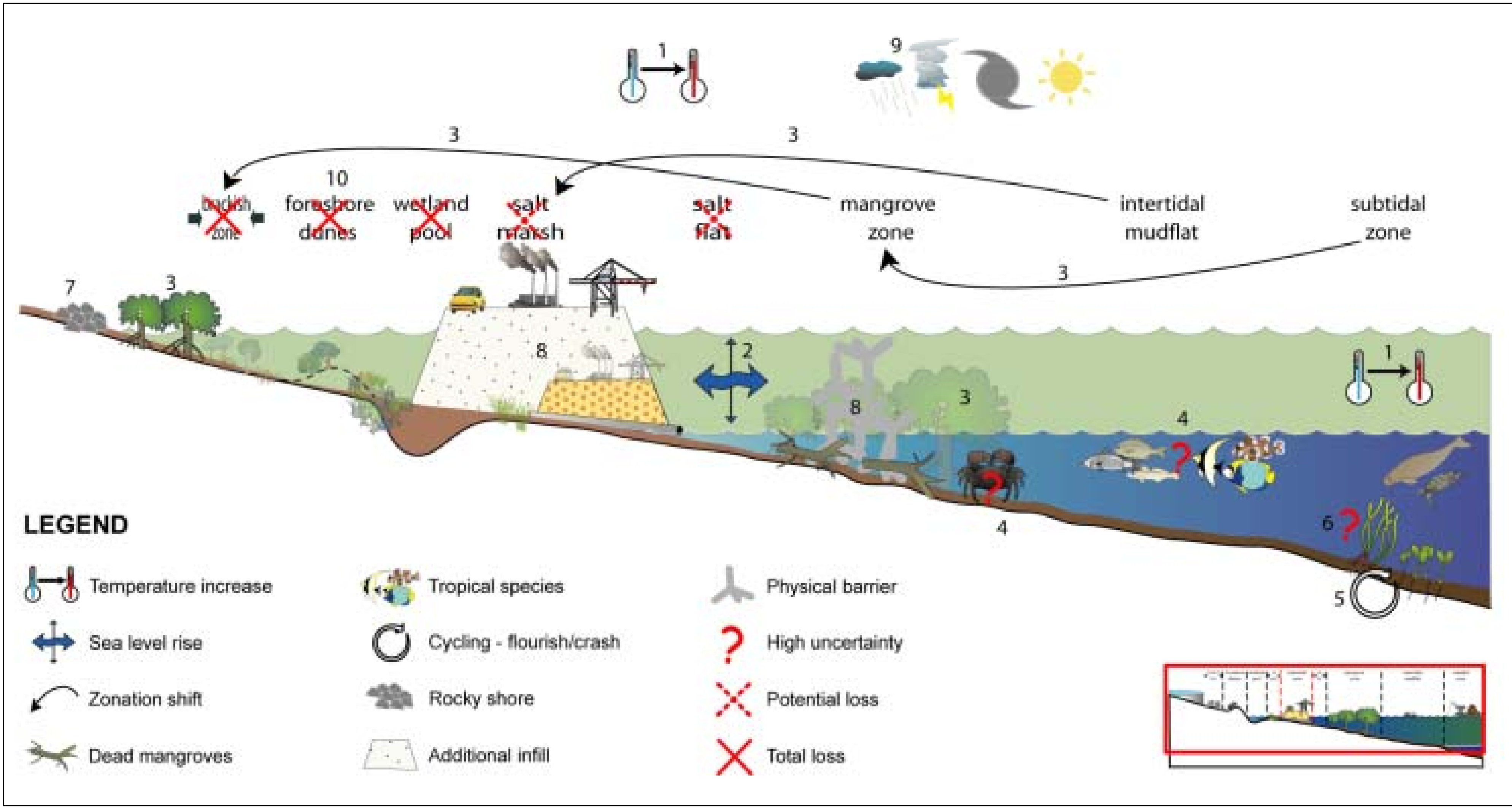

Diagram 7.12. Pressures associated with climate change and sea level rise

7.4 Summary

While there is some information available for the Port Curtis intertidal area, much of the information is based on small-scale or one-off projects. It is therefore difficult to accurately determine processes and pressures for any given aspect or area of the intertidal system. However, collectively, all the information provides a relatively good appreciation of what is happening within Port Curtis. As further information becomes available it can be incorporated into conceptual models such as these, eventually providing a complete picture of the area. Conceptual models also provide the opportunity to identify gaps in our knowledge base, and through time it is anticipated that these gaps will be filled through research prioritisation.

At this stage it appears that the Port Curtis intertidal wetland system is in relatively good health, despite the level of industrialisation in the area. However, there is also much relevant information that is lacking from our current knowledge of the system. For example, nothing is known of the ecology and biodiversity associated with sandy and rocky shores or saltflat communities. All of these systems have essentially been ignored, while mangroves, water quality and benthic communities in soft sediments have been relatively well studied. As yet it is still not clear how important areas such as the saltflats may be to the functioning of the intertidal wetlands.

The lack of baseline information for almost all aspects within Port Curtis has made establishing any causal links impossible. While the intertidal and marine system appears to be functioning normally at present, it is unknown whether the system may have been different prior to European colonisation, or even prior to the increase in industrialisation since the 1960s. For instance, the biomass of prawns may have been larger in the early 1900s than it is now. Critical knowledge gaps identified for further study in the Port Curtis intertidal wetlands are further discussed in the final chapter of this report.

7.5 Glossary of terms used in conceptual models

Algae	A group of several plant-like phyla, including the red, green and brown algae, and diatoms. Algae lack true stems, roots and leaves.
Anoxic	The condition of oxygen deficiency or absence of oxygen. Anoxic sediments and anoxic bottom waters are commonly produced where there is a deficiency of oxygen due to very high organic productivity and a lack of oxygen replenishment to the water or sediment, as in the case of stagnation or stratification of the water body.
Autotroph	An autotroph (or producer) is an organism that makes its own food from light energy or chemical energy without eating.
Benthic	Animals dwelling on the bottom of a water body. These organisms inhabit the sediment on lake, river or ocean bottoms, as well as the sediment in marshes, tidal flats and other wetlands.
Bioturbation	The mixing of sediments by living organisms such as worms, clams or arthropods that make burrows in soft sediment.
Climax community	The stage in community succession where the community has become relatively stable through successful adjustment to its environment.
Consumer	Any organism that cannot produce its own food and must therefore get its energy by eating, or consuming, other organisms.
Desiccation	To lose water and dry up.
Detritus	Dead or decaying organic matter (including animal and plant material).
Ecotone	A transition area between two distinct, but adjoining, communities or ecosystems.
Epiphyte	A plant growing on, but not parasitic on, another plant.
Estuary	A semi-enclosed body of water which (at least sometimes) has a free connection to the open sea and within which sea water is measurably diluted by fresh water derived from land drainage.
Evapotranspiration	Loss of water from the soil both by evaporation and by transpiration from plants.
Flocs	Clumps of bacteria and particulate impurities or coagulants that have come together and formed a cluster.
Infauna	Aquatic animals that live in the substrate of a body of water, especially in a soft sea bottom.
Macrobenthic	Animals dwelling on the bottom of a water body that are equal to or greater than 0.5 mm in size.
Microphytobenthos	Microscopic, photosynthetic eukaryotic algae and cyanobacteria that grow in or on mud in aquatic habitats.
Novel chemicals	New chemicals to an environment or system. Can be contaminants, including nutrients.

Plankton blooms	Blooms of small, microscopic algae (phytoplankton) or animals (zooplankton) in aquatic systems.
Pore water	Water between sediment grains.
Post-larval	The life stage in many invertebrates and fishes that follows the larval stage (larvae = early life history stage that is unlike its adult form and must metamorphose before assuming adult characteristics).
Primary producer	An organism capable of using the energy derived from light or a chemical substance in order to manufacture energy-rich organic compounds, mainly green plants and algae.
Recruitment	The process where a juvenile marine species has survived long enough to become a part of (i.e. recruited into) a population or an exploitable segment of the population.
Secondary producer	A heterotroph; an organism which requires organic matter and energy from the biotic environment.
Subtidal	The portion of marine environments that lies below the level of mean low water for spring tides. Normally covered by water at all stages of the tide.
Succession	The natural replacement of one plant or animal community by another over time in the absence of disturbance.
Trophic	Of or involving the feeding habits or food relationship of different organisms in a food chain.
Turbidity	The cloudy or muddy appearance of a naturally clear liquid caused by the suspension of particulate matter.

7.6 References

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Chapter 8. Ecological conclusions and their implications for Port Curtis

Rod M. Connolly and Michelle A. Dawson

8.1 Introduction

The results presented in this report provide improved knowledge of the patterns and processes in the Port Curtis intertidal wetland environment relevant to government and management agencies, industry, research institutions and the general community. The coastal mapping provides a definitive inventory of intertidal habitat distribution. The conceptual diagrams provide an understanding that can be built upon as further information about processes and pressures in Port Curtis is gathered; they also highlight general concepts that may be applied to other subtropical estuaries. The results of this project have allowed us to identify critical knowledge gaps for Port Curtis (see Section 8.2). Furthermore, a synthesis of the key outcomes from each chapter, discussed in Section 8.3, highlights the importance of different intertidal wetland habitats in the study area. The report has brought together information about intertidal wetlands in Port Curtis that will assist in selecting future coastal management options, and will facilitate subsequent assessment of the ecological values of these habitats.

8.2 Knowledge gaps

Research into the ecological roles of the intertidal wetlands in Port Curtis is important for this growing coastal port, to assist in quantifying the social, economic and environmental values of the area. The results presented in this report provide good baseline data for previously little understood processes. We have, however, identified remaining knowledge gaps that hinder a full understanding of the biophysical and geochemical processes that underpin the Port Curtis system.

8.2.1. Specific research areas

Knowledge gaps identified from this project are listed below under the headings used in Chapter 7. Points under each heading have not been prioritised.

Tides

- Periodic inundation of wetlands provides connectivity between the marine environment and brackish pools, providing the pools with fresh nutrients and food resources such as plankton, while providing some level of flushing within the pools. Although within the dry tropics this connectivity presumably plays some role in the overall mangrove-based intertidal ecosystem, the effect of this connectivity is still not well understood and needs further investigation.

Freshwater flows

- The effect of freshwater flows on downstream communities within the subtidal zone is largely unknown. Hegerl and Tarte (1974) suggest that even minor changes in the volume of freshwater runoff into the tidal wetlands of the Capricorn area may result in unexpected changes in flora and fauna.
- Transportation of sediments and other particulate matter deposited in mangroves and other upper intertidal habitats may be driven via the tidal system rather than freshwater runoff; however this has not been quantified.
- The shelter provided by mangroves facilitates the deposition of sediments and flocculated material, and consequently the associated nutrients. The importance of this process in encouraging sediment/nutrient deposition in the intertidal zone and preventing nutrient passage to the open waters of the estuary and beyond requires investigation.

Trophic links

- The process by which carbon is transferred from seagrass to secondary producers is not known. For example, is the carbon transferred to detrital systems before passing onto higher consumers, or do the higher consumers eat the seagrass directly? Algae epiphytic on seagrass may also play a significant role in the transfer of carbon.
- Carbon dynamics on the saltflats are not known at this stage, but it is assumed that the majority of production occurring in this area remains *in situ* due to the low rate of use by mobile consumers and the intermittent inundation from the tides. The extent of export from cyanobacterial blooms also remains to be tested.

- It is possible that carbon from lower intertidal areas may end up in the upper intertidal areas, such as the saltflats or wetland pools, via trophic interactions. This 'inwelling' of organic matter has been recorded elsewhere but no evidence for or against such a pathway has been gathered in Port Curtis.
- The rate of commercial and recreational harvest of certain key species (e.g. mud crabs) may result in cascading effects on lower trophic levels. This has been an important theoretical consideration elsewhere but has not been addressed in Port Curtis.

Storms

- The impacts of major storm events on the Port Curtis intertidal ecosystem is considered likely to be very important, but as yet there is little information on this topic. Aspects such as the effects of storms on seagrass health and colonisation would be particularly important.

Droughts

- The impact of droughts on intertidal communities such as seagrass meadows is unknown. Interactions occurring within the seagrass meadows are not well understood, and it is possible that the effects of drought on seagrass itself and on grazing animals may interact to produce currently unpredictable results.
- Primary production within mangroves may decrease during a drought, and prolonged droughts might ultimately lead to changes in mangrove distributions, but neither aspect has been quantified.

Infill

- Infilling affects tidal connectivity of high intertidal habitats. The effects of altering levels of connectivity between, for example, saltflats and open estuarine waters, are unknown. Likely effects of infill would, for example, be on production and species distributions on saltflats.
- By increasing the area of developed land, runoff from agricultural, industrial and urban sources is increased, concentrating flow into point-source outfalls rather than natural diffuse runoff. Sheeting flow of fresh water over saltflats and saltmarsh and filtration through mangrove systems is reduced, potentially removing the biological cues for secondary processes that may occur as a result of the freshwater inflow.

Dredging

- A potential increase in the level of estuarine contamination may be observed after dredging operations if many years worth of sediment-linked contaminants are resuspended over a short period. To date, no causal link has been established between dredging and contamination levels within Port Curtis, but monitoring should be used to assess this potential link given increasing levels of construction and development in the area.

Barriers, dams and weirs

- The effects of barriers and dams on mangrove distributions and productivity have not been studied in Port Curtis (Johnson & Rogers 2004). Reduced freshwater flows have resulted in mangroves extending their distribution further upstream in estuaries in other parts of the world, and this effect would be likely in Port Curtis.
- The effects of barriers and dams on saltmarsh are unknown in Port Curtis, but any reduction in the supply of fresh water would be expected to be detrimental to saltmarsh vegetation because of the increased salinity of pore water in sediments.

Trends in recreational fishing catches

- As highlighted in Chapter 5, fish catch rates inside Port Curtis are apparently declining more quickly than those at Cape Capricorn. As habitat modification has been historically much greater within Gladstone harbour, this relationship is worthy of further investigation.
- The correlation found between fish catch and macrobenthos in Port Curtis indicates an ecological link between them, possibly reflecting a decline in overall estuarine productivity. As an important region for both recreational and commercial fisheries, further investigation of this trend in Port Curtis is warranted.

8.2.1. Overall gaps

Overall, it appears that the Port Curtis intertidal wetland system is in relatively good health, despite the level of industrialisation in the area. However, relevant information about the system and its responses to potential changes in tidal connectivity and freshwater flows is lacking. Additionally, almost nothing is known of the ecology and biodiversity associated with sandy and rocky shores on the more exposed coasts of Port Curtis. We have also highlighted the paucity of information about the very extensive saltflat habitats that are likely to be adversely affected by immediate impacts resulting from human activities and longer-term climate changes. Given the importance of seagrass ecologically, an assessment of sediment movement and its relationship with seagrass extent or decline would be valuable. This could be usefully included as part of an overall study of hydrodynamics and sediment movements and distribution within the Port.

8.3 Conclusions

8.2.1. Specific findings

Coastal mapping

A mapping survey of mangroves, saltmarsh, saltflats, mudflats, seagrass and intertidal rocky substrate in Port Curtis combined data from Landsat satellite imagery, aerial photography and field inspections. The intertidal wetlands of Port Curtis are characterised by strong zonation. Saltflats without vegetation (landward of mangroves) cover 18% of the intertidal area. At the edge of these saltflats, where conditions are less saline, samphires and saltmarsh grasses occur, though they occupy only 4% of the intertidal area. Mangroves cover 31% of the area. Unvegetated mudflats (seaward of mangroves) cover 25% of the area. Seagrass meadows were estimated to cover 20% of the area, although this probably overestimates seagrass cover for much of the year. Cover within defined seagrass areas is patchy and highly seasonal, and the estimate was made at the period of peak seagrass cover.

Much of the seagrass in Port Curtis is close to existing or proposed port infrastructure and dredged channels. Although the seagrass is apparently healthy, the location of meadows leaves them vulnerable to direct impacts of future port infrastructure developments such as wharves, breakwaters and reclamation. The seagrass is of regional importance as one of the few large areas of seagrass in central Queensland, with the closest other areas more than 170 km to the north and south.

The Shoalwater Coast bioregion (which includes the Curtis Coast) has the second largest proportion of saltmarsh/saltflats of any estuarine area in Queensland, presumably because of its aridity (average annual rainfall only slightly over 1000 mm). Over 1600 hectares of tidal wetlands, particularly saltflats, were lost from Port Curtis between 1941 and 1999, due to large-scale reclamation (infilling) for industrialisation, urbanisation and port development.

Dietary analysis of juvenile fish (Chapter 6) shows that some species of fish (e.g. mullet) feed on saltflats, so the loss of this habitat could have an adverse effect on fisheries sustainability.

The habitat maps produced as part of this project provide a detailed summary of the distribution of the intertidal wetlands in Port Curtis, and can play an immediate role in planning and decision making in the area.

Physicochemical characteristics of surface and subsurface sediments

The chemical characteristics of mudflat, mangrove, saltflat and seagrass sediments were compared to determine associations between sediment parameters and habitat types. Physicochemical parameters were strongly correlated with, and probably determine, zonation of intertidal vegetation. Porewater salinity was the most important parameter, maintaining the extensive hypersaline saltflats landward of the main mangrove stands. At the land–sea margin, freshwater runoff dilutes porewater salinity sufficiently to allow a narrow strip of mangroves to take over from saltflat habitat, but not so much as to allow colonisation by less salt-tolerant terrestrial trees. Any reduction in freshwater flows resulting from increased development, land infill or climate change would alter the vegetation composition in Port Curtis.

Trends in recreational fishing catches

Recreational fishing is very popular and has a very high participation rate in Gladstone. The state of recreational fisheries is one indicator of environmental health and ecological productivity. Catch rates of whiting are lower in Port Curtis than at Cape Capricorn situated immediately outside the Port. Both locations show a decline in catch rates over time, but over the last ten years this trend has been more obvious inside the Port than at Cape Capricorn. No significant trend was established for offshore catches.

The investigations provide evidence that freshwater flows are important to the productivity of Port Curtis. Years of large flow tend to have higher benthic invertebrate productivity, resulting in higher growth rates in fish such as whiting, which are caught earlier in their life cycle than they would otherwise be. This relationship between freshwater flows and fisheries productivity has been shown in several other places in the world, although the relationship does not hold true in all situations. The influence of flow on catch rates probably does not surprise local stakeholders but the immediacy of the effect (i.e. absence of a lag effect) is surprising. Reductions in freshwater flows as a result of increased water extraction for industry or urban use, or as a result of climate change, are likely to have an adverse effect on fisheries sustainability.

Distribution and assemblage composition of demersal fish in shallow near-shore waters of Port Curtis

A trawl survey of 105 intertidal and shallow subtidal sites showed that fish assemblages of Port Curtis are diverse, with 88 species of fish recorded. The catch was dominated numerically by two small schooling species, ponyfish (*Leiognathus equulus*) and herring (*Herklotsichthys castelnaui*), common in inshore coastal waters elsewhere in tropical and subtropical waters of Australia. Fish assemblages change gradually from the more estuarine waters of The Narrows to open coastal waters. In addition, assemblages were distinguishable in different habitats, particularly seagrass meadows, and in varying water depths.

Trophic importance of high intertidal habitats to fish in Port Curtis

Although it is known that seagrass is important to estuarine productivity (Connolly *et al*. 2005), until now the precise role of seagrass in Port Curtis has been unclear. The current research provides more locally specific evidence of the role of seagrass (Chapters 5 and 6). Seagrass was found to support a unique fish assemblage, thus promoting local biodiversity. Studies using stable isotope analysis to trace energy pathways and nutrient cycling demonstrated that seagrass is also important in food webs beyond the meadows themselves. The food webs that sustain the many economically important fisheries species caught predominantly over mudflats (e.g. whiting) or in mangrove-lined creeks (e.g. mud crabs) rely largely on organic matter produced in seagrass meadows. Seagrasses cover a smaller area of the Port than do mudflats or mangroves (Chapter 2), yet contribute more to the trophic requirements of animals living over mudflats. This is contrary to the early ecological theory that mangroves drive subtropical estuarine food webs. While mangroves may be important to the animals living in the mangroves (Bouillon *et al*. 2002), there is no evidence of a trophic role for fisheries species occurring in other habitats in Port Curtis. It is important to note, however, that mangroves do provide other ecosystem benefits by stabilising and oxygenating sediments, and providing habitat for juveniles of economically important species such as banana prawns.

Over the last decade, a paradigm has emerged that in areas dominated by large mudflats, benthic microalgae are important in food webs. This view is based partly on the higher rate of productivity of benthic microalgae and partly on direct stable isotope analysis (Middelburg *et al*. 2000). Such a role for microalgae has been suggested for the Fitzroy estuary just to the north of The Narrows. However, in the subtropical waters of Port Curtis, microalgae do not appear to play a major trophic role for the fisheries species studied. A less important role for microalgae in food webs has been reported in other subtropical (Melville & Connolly 2005) and temperate (Connolly *et al*. 2005) bays in Australia.

A conceptual understanding of ecological patterns and processes maintaining coastal wetlands in Port Curtis was developed in interviews with scientists and other stakeholders. This led to a series of conceptual diagrams that summarise current knowledge about intertidal wetlands in this subtropical port. Five attributes or processes were identified as being of great importance: 1) tidal water movement, 2) freshwater runoff, 3) trophic (feeding) links, 4) severe storms and 5) droughts. The five top pressures impacting the wetland ecosystem were identified as: 1) land infill, 2) dredging, 3) barriers and dams, 4) contaminants (including plant nutrients) and 5) climate change.

8.2.1. Overall conclusions

The proper management of Port Curtis and its estuarine wetlands depends on the quality of available information. This report can assist in the identification of management measures, such as protocols to limit dispersal of sediments during dredging. The habitat maps generated as part of this work provide a detailed summary of the distribution of the intertidal wetlands in Port Curtis, and can play an immediate role in planning and decision making in the region. The trade-off between port development and management for biodiversity, at sites where development is designated to occur, requires an understanding of the importance of the wetland ecosystems. This report includes scientific criteria required for determining the value of the intertidal wetlands of Port Curtis, and can direct the focus for future research aimed at better informing management decisions about the relative significance of the wetland habitats.

8.4 References

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