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Impact of urbanisation on coastal wetlands

a case study of Coombabah Lake, South-east Queensland

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The impact of urbanisation on coastal wetlands: a case study of Coombabah Lake, southeast Queensland

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Contents

List of figures

List of figures (continued)

List of tables

List of tables (continued)

Executive summary

Introduction

This study investigates the impact of urbanisation on coastal wetlands through a detailed case study of Coombabah Lake in southeast Queensland. Coombabah Lake has an area of about 460 hectares and is surrounded by tidal wetlands dominated by mangroves and salt marshes. It is fringed on the seaward and landward sides by tidal mudflat and melaleuca forests, respectively. Significant modification of the lake's catchment has occurred in the last three decades, replacing significant areas of natural bushland by urban and industrial developments.

Coombabah Lake embodies the type of urbanisation pressure experienced by wetlands in fast-developing coastal areas. With an additional one million people expected to migrate to the coastal region of southeast Queensland in the next 20 years, increasing urbanisation is inevitable and will affect the capacity of the region's coastal wetlands to sustain their beneficial ecosystem services.

The Coombabah Lake case study

The current study characterises the present condition of Coombabah Lake in terms of water exchange, circulation, sediment and pollutant resuspension and accumulation. It also assesses the impact of local urban runoff on ecosystem structure and function. A functional model (Figure 1-1) is presented to demonstrate the general impact of urbanisation on coastal wetland structure and function.

The four components of the study are: (1) characterising and assessing the condition of the surface sediment (0–5 cm) in the lake using a number of physical, chemical and biological parameters; (2) investigating the tidal inundation pattern in the lake in relation to the provision of animal habitats; (3) understanding the water–sediment interaction in the lake, with particular focus on sediment resuspension; and (4) assessing the impact of urban influx on the lake through the use of selected biotic indicators.

This report

The full report presented here consists of a comprehensive introduction followed by technical reports on all research activities carried out as part of the study. Each technical chapter is introduced by a brief chapter overview, describes in detail the approach and methods used, discusses results of the research and concludes with a summary of the findings and their implications for Coombabah Lake and beyond.

Findings

Results suggest that surface sediments in Coombabah Lake are typical of Australian estuaries and there is presently no significant concern about the level of major pollutants in the lake except for local elevated levels of two heavy metals, nickel and arsenic. The lake has, however, a low buffering capacity for metal influx, making it vulnerable to increased contamination.

Radioactive carbon dating of shells suggests that sedimentation rate in the lake has been extremely slow and has not been greatly accelerated by existing urbanisation. Despite the shallow depth of the lake, wind- and tide-induced sediment resuspension also appeared to be weak, resulting in low sediment transport and export into the Coomera estuary. Sediment export is probably only effective during major flooding events. The sediment transport pattern therefore facilitates gradual infilling, albeit currently only at a slow rate. The lack of an effective sediment export mechanism also implies that material entering the lake from the upland catchment would be retained in the wetland, pointing to the need for careful management of future urbanisation and other processes that may generate increased loads of sediment and other contaminants.

Tidal flushing in Coombabah Lake was not even, but comprised five distinct patterns in areas that are flooded at different trigger tide levels. These areas also have different durations of inundation. Local micro-topography is an important factor controlling the hydrologic regimes.

Levels of the biotic indicators also suggest that Coombabah Lake is generally of a good condition. There are strong signs, however, that local influx sources are shaping the condition of the lake environment. Despite the significant urbanisation occurring at the Helensvale and upper Coombabah Creek subcatchments, these influx sources generally result in weaker disturbance to local habitat condition than the discharge at the creek near the Coombabah Sewage Treatment Plant. Indicators that are more responsive to tidal changes in nutrient regime during the dry–wet transition suggest that adverse impacts can be detected around local urban influx sources at an extent of hundreds of metres during the wet period. Indicators that are reflective of average long-term conditions also support the presence of some local negative effects of urban discharge on the lake biota.

Implications for Coombabah Lake

The study reveals that although Coombabah Lake is currently relatively healthy in the face of significant urbanisation, the system appears to be quite vulnerable to future changes in its catchment. The tendency of the lake to retain rather than export particles coming from its catchment, the weak buffering capacity for increased metal input, and the presence of local negative impacts around urban influx points that are exacerbated by rainfall runoff events all point to the fact that careful planning, monitoring and management are crucial to the capacity of the lake for sustained beneficial ecosystem services.

Further development

This study sets the baseline for the condition of Coombabah Lake in terms of the biophysical environment and the level of biotic indicators relevant to the structure and function of the wetland. The spatially explicit sampling design and the use of selected biotic indicators also provide a model for repeated analysis of the trends and patterns in the future.

Although the impact of urbanisation can to an extent be inferred from local spatial patterns of sediment characteristics and trends in the biotic indicators, time-series measurements of these parameters would allow a better assessment of the importance of urbanisation as a driver of change in the wetland.

Snapshot measurements of water quality are currently implemented in Coombabah Lake by the Gold Coast City Council. Water quality is nevertheless highly variable, and provides little time-integrated indication of habitat condition. In contrast, sediment and biological sampling such as those conducted in this study provide better time integration of condition, as well as more proximal indication of potential impact.

Continued monitoring of the water and sediment quality and ecological health of the lake will be necessary to ascertain changes over time and response to specific events or management actions. Similar monitoring programs are currently in place in Moreton Bay (e.g. the Healthy Waterways Ecosystem Health Monitoring Program), and there are obvious benefits to be gained through integrating future monitoring work in Coombabah Lake with these broadscale programs.

Chapter 1

Introduction

The function and value of coastal wetlands

Wetlands are biologically diverse and productive transitional areas (or 'ecotones') between land and water, characterised by shallow water overlying waterlogged soil and interspersed with submerged or emergent vegetation. By occupying zones of transition between terrestrial and marine ecosystems, coastal wetlands—including *Melaleuca* swamps, salt marshes, mangroves, intertidal mudflats, seagrass beds and shallow subtidal habitats—are the interface of the coastal landscape. It is generally understood that wetland formations are determined by the cumulative interactions of hydrology, landscape position, sediment dynamics, storm-driven processes, sea level rise, subsidence and colonisation and disturbance by animals (Varnell *et al.* 2003).

Coastal wetlands have been suggested to offer many important ecosystem services (Woodward & Wui 2001), but few of these have been demonstrated, including the highly controversial 'outwelling' role (Odum 1980), which hypothesises that coastal wetlands export large percentages of their production to support offshore secondary production (Lee 1995). Being productive and often spatially diverse habitats, wetlands fulfill important functions such as providing habitats for flora and fauna including migratory birds (Penthick 1984; Nakamura *et al.* 1997; Messina & Connor 1998; Stumpf & Haines 1998; Dierschke *et al.* 1999; APMWCC 2001; Ishikawa *et al.* 2003) and helping to moderate water quality (Faulkner 2004). Other migratory species, including fish, turtles and cetaceans, also utilise inshore wetlands. Coastal wetlands such as mangroves, salt marshes, intertidal mudflats and seagrass beds produce a large variety of food to consumers, with system net primary productivity levels of mangroves and salt marshes often exceeding 2000 and in some cases reaching 4000 grams dry weight per square metre per year (Alongi 1998). In addition to providing habitat and food sources for associated organisms, wetlands may also support commercial and recreational fisheries through their role as nursery habitats (Lee 1999; Clynick & Chapman 2002; Ansari *et al.* 2003) and deliver several direct and indirect services to the local population (Bird 1984).

Wetlands may also act as a buffer between land and sea as they prevent erosion, reduce currents, attenuate waves and encourage sediment deposition and accretion (Bird 1984). Analysis of the recent Asian tsunami suggests that there may be an inverse relationship between mangrove presence and tsunami damage (e.g. Dahdoub-Guebas *et al.* 2005).

Safeguarding such vital roles requires resource management strategies that are often at odds with increasing human development pressures (Stumpf & Haines 1998). In contrast to the high ecological value placed on coastal wetlands, they are often utilised for a number of destructive and consumptive uses such as waste dumping, land reclamation, aquaculture ponds and dredging for navigational channels and marinas. Such activities have resulted in the recent rapid loss of coastal wetland habitats such as salt marshes and mangroves. For example, more than half of the mangrove forests present in many southeast Asian countries in the 1960s have already been lost to some form of development (Wilkinson *et al.* 1994), and the trend is continuing (Ewel *et al.* in review). Ironically, such destruction is concomitant to an increasing awareness of the ecosystem services provided by coastal wetlands. One important emerging concept is that coastal wetlands deliver beneficial ecosystem services such as supporting fishery production, as a complement of habitats rather than in isolation (e.g. Lee 2004), which has strong implications for their conservation and management. In a meta-analysis, Woodward and Wui (2001) demonstrated that a site-specific approach may be necessary in evaluating wetland services. Coastal wetlands also provide significant non-consumptive services such as tourism and conservation of biodiversity resources but the value of these services is often difficult to quantify.

Urbanisation and its impacts on coastal wetlands

Urbanisation is often referred to as either the degree of or increase in urban character or nature, and may refer either to a geographical area combining urban and rural areas or to the transformation of areas into greater urban development. The term may be used to describe a condition at a specific time, namely the proportion of total population or area in urban localities or regions, or the increase of this proportion over time.

By definition, urbanisation involves the replacement of natural habitats by built-up areas that support human inhabitation and its associated activities. It is a major cause of loss of coastal wetlands around the world.

Urbanisation exerts significant influences on the structure and function of coastal wetlands, mainly through modifying the hydrological and sedimentation regimes and the dynamics of nutrients and chemical pollutants. Natural coastal wetlands are characterised by a hydrological regime comprising concentrated flow to estuarine and coastal areas during flood events and diffused discharge into groundwater and waterways during non-flood periods. Through increasing the amount of impervious areas in the catchment, urbanisation results in a replacement of this regime by concentrating rain runoff. Quality of runoff is also modified in urban areas as loadings of sediment, nutrients and pollutants are increased. While the effects of such modifications on the biota and the physical environment have been relatively well studied, there is to date little information on their impact at the ecosystem level. Methodological issues—such as a lack of sufficient replication at the whole-habitat level, the lack of suitable indices of urbanisation and the lack of tools for assessing hydrological connectivity—have to be overcome to allow the effects of urbanisation to be assessed at the ecosystem level.

Globally, wetland ecosystems are under pressure from rapidly increasing urban populations in coastal areas (Ehrenfeld 2000). For example, coastal counties (i.e. those within 80 km of the coast) make up only 13% of the land area of the continental United States, but this area encompasses 51% of the population (Rappaport & Sachs 2003). Within Australia, about 84% of the population live within the coastal region (Australian Bureau of Statistics 2002). Outside Australian capital cities, most population growth has occurred in coastal areas such as the city of Gold Coast in Queensland, which increased by around 240 500 people between 1986 and 2003 (Australian Bureau of Statistics 2002; Office of Urban Management 2004). The continuing population growth within coastal regions ensures that there will be ongoing impacts on coastal wetland ecosystems (Callaway & Zedler 2004).

Hydrological and sedimentation regimes are the main physical drivers in coastal wetlands. Both are often significantly modified by human activities and introduce additional drivers such as pollutants, exotic species, harvesting of biomass and direct habitat loss. The impact of urbanisation on coastal wetlands is initially and mainly through alteration of hydrological and geomorphological processes. Structural and functional ecological changes then follow. The intensities of these disturbances vary along different spatial and temporal scales (Lindegarth & Hiskin 2001). Impacts caused by one type may be modified by other disturbances within the system, creating complex interactions between them. For example, as a result of urbanisation-led changes in sediment loads, water quality, aquatic systems and associated organisms within affected areas may be exposed to elevated concentrations in trace metals, pesticides, hydrocarbons and nutrients (Horner 2000; Burton *et al.* 2004). Additionally, other types of disturbances in areas affected by urban development may include: structural and hydrological modifications (including altered stormwater runoff, drainage and filling characteristics); ecological modifications; recreational activities; the introduction of exotic species; and resulting effects from increased sedimentation loads (Ehrenfeld 2000; Horner 2000; Lindegarth & Hiskin 2001). Other disruptions include those to wildlife migration routes.

The main impacts of urbanisation on coastal wetlands are discussed in more detail in the following sections of this chapter.

Hydrological and geomorphological impacts

Table 1-1 summarises the potential effects of urbanisation on wetland hydrological and geomorphological processes.

Table 1-1. Impact of urbanisation on wetland hydrology and geomorphology (modified from Ehrenfeld, 2000)

Hydrology	Impacts
	• Decreased surface storage of stormwater results in increased surface runoff (and, in turn, in increased surface water input to wetland). • Increased stormwater discharge relative to base flow discharge results in increased erosive force within stream channels, which leads to increased sediment inputs to recipient coastal systems. • Changes occur in water quality (e.g. increased turbidity, increased nutrients, metals, organic pollutants and/or decreased oxygen concentration). • Culvert, outfalls, etc. replace low-order streams; this results in more variable baseflow and low-flow conditions. • Decreased groundwater recharge results in decreased groundwater flow, which reduces base flow and may eliminate dry-season stream flow. • Increased flood frequency and magnitude result in more scour of wetland surface. • Increase in range of flow rates (low flows are diminished; high flows are augmented) may deprive wetlands of water during dry weather. • Greater regulation of flows decreases magnitude of spring flush.
Geomorphology	**Impacts**
	• Decreased sinuosity of wetland/upland edge reduces amount of ecotone habitat. • Decreased sinuosity of river channels results in increased velocity of stream water discharge to receiving wetlands. • Alterations in shape and slopes (e.g. convexity) affect water-gathering or waste-disseminating properties. • Increased cross-sectional area of stream channels (due to erosional effects of increased flood peak flow) increases erosion along banks.

Alteration of the hydrological regime

The hydrological regime is a key determinant in the morphology, species distribution, productivity, sedimentation rates, pollutant transport and nutrient cycling and availability of coastal wetlands (Owen 1995; Hughes *et al.* 1998; Keddy 2000). Conversely, the structure of wetland systems—for example, the type of vegetation cover—is also critical in determining the hydrological balance (Hughes *et al.* 1998). The structure and function of wetland ecosystems are determined primarily by the hydrological regime and its effect on sediment geochemistry (Ellery *et al.* 2003). The tolerance ranges of species to the frequency, depth and duration of inundation exert strong controls on the distribution of flora and fauna within wetlands (Lui *et al*. 2002; Ellery *et al.* 2003). Elements of hydrological management include long-term gradual changes (e.g. climate change and projected sea-level rise) and sudden changes resulting from human interference (e.g. hydraulic modification of tidal flow) (Hughes *et al.* 1998). Direct hydrological changes in wetlands commonly occur as a result of urban development (Penthick 1984; Horner 2000; Lindegarth & Hiskin 2001).

Structural and hydrological modifications (including altered stormwater runoff, drainage and filling characteristics) may result in elevated concentrations of trace metals, pesticides, hydrocarbons, sediments and nutrients within the receiving wetland (Nakamura *et al.* 1997; Horner 2000; Lindegarth & Hiskin 2001). Physical changes to landscapes, as a result of massive land movement associated with urban infrastructure construction, also cause geomorphological changes both in the wetlands and adjacent catchment (Nakamura *et al.* 1997).

Wetland hydrology involves: (i) groundwater flow; (ii) ground surface flow; (iii) water column flow (tidal water); and (iv) evapotranspiration (Varnell *et al*. 2003). These functional water groups within wetland environments interact with local sediment and nutrient supply to control basin morphology (Varnell *et al*. 2003). Hydrological components are further interactively affected by factors such as surface roughness, topography, dominant vegetation type, pattern of rainfall and tidal range (Nuttle & Harvey 1995; Hughes *et al*. 1998; Darboux *et al*. 2001; Bendjoudi *et al*. 2002). Additionally, wetland hydrology is affected by such factors as land uses within the catchment, wetland–to–watershed area ratios and system characteristics such as sediments, bathymetry, vegetation and inlet and outlet conditions (Horner 2000).

Conversely, the relative importance of these water components in a wetland's hydrological regime would determine the degree of connectedness with adjacent nearshore and terrestrial habitats. Well-connected alluvial corridors within catchment systems characterised by complex surface and subsurface structures provide a wide range of aquatic habitats (Poole *et al*. 2002). Floodplain surface topography also influences the location and composition of vegetative communities on a flood plain. Furthermore, as a control on groundwater and surface-water exchange, fluvial corridor surface and subsurface morphology can influence biodynamics in fluvial systems by influencing solute transport, carbon and nutrient cycling and nutrient availability in the channel and hyporheic zone (Poole *et al*. 2002). Floodplain geomorphology is therefore the template upon which hydrologic function evolves and the resulting habitat diversity and biocomplexity depend (Poole *et al*. 2002).

Elements of water quality that are affected by changes in wetland hydrology include nutrient transformations, availability, deposition and organic material flux within the system (Whitehouse *et al*. 2000). Net circulation is responsible for exporting organic matter from intertidal mangrove wetlands (Wolanski 1992). Greater surface runoff is also likely to increase the velocity of inflow to wetlands, which can potentially disturb the resident biota and scour substrata. Sediment retention within coastal wetlands and estuaries is directly related to flow characteristics, including the degree and pattern of channelisation, flow velocities, and storm surges (Horner 2000).

Impact of urbanisation on the hydrological regime

Hydrological change is the most direct impact of urbanisation and strongly influences water quality, as well as the hydrodynamic variables within the system. In addition to changes in nutrient and sediment levels, more subtle changes such as the ratio of particulate to dissolved organic matter and its lability may be affected (Hopkinson & Vallino 1995). Urbanisation typically increases runoff peak flows and total flow volumes and damages water quality and aesthetic values (Horner 2000), through conversion of wetland soil into impervious surfaces. Alternatively, dam construction and water extraction due to increasing water demand in urbanised areas also affects coastal wetlands by altering the level and frequency of environmental flow.

Hydrologic changes can make an area more vulnerable to pollution as increased water depths or frequencies of flood can distribute pollutants more widely. Changes in hydrological flows generally result in sedimentation instream and decreased depths to the extent that vegetation, especially exotics, invades shallower sections. While floodwaters distribute pollutants more widely, they also result in increased dilution of pollutants. It is the relatively smaller but more frequent stormwater flows which previously infiltrated soils that now regularly disperse pollutants without the benefit of dilution.

Urbanisation usually involves the conversion of natural habitats to land uses with impervious surfaces, for example, pedestrian paths and roads (Faulkner 2004). These surfaces block the infiltration of precipitation and cause changes in hydrology, which degrade downstream ecosystems. Impervious surfaces channel sediments and pollutants directly into drainage networks, thereby increasing stormwater runoff into receiving wetlands (Faulkner 2004). In addition, pollutants bound to sediments and organic matter (Connell *et al.* 1999) are distributed through these artificial transport systems and into wetland habitats. This impact will be dealt with in greater detail later.

As changes in surface and subsurface water flow become more common with increased land-use change, more information is needed regarding the factors that influence survival and recovery in sensitive forested wetland areas (Ernst & Brooks 2003). Unfortunately, because of the complex nature of the wetland–watershed relationship, there is still a great deal of uncertainty over the hydrologic budgets and functions of different wetland types (Owen 1995). Most of the existing hydrologic studies of wetlands have been conducted in relatively simple systems for which the components of hydrologic budgets can be estimated (Owen 1995). These include diked impoundments and wetlands with a distinct inlet and outlet. Lake-edge or creekside wetlands are inherently more difficult to study because of the difficulty of accurately measuring sheet flow across the wetland surface into or out of the stream or lake. Similarly, shallow, horizontal groundwater flow through substrata with highly variable hydraulic conductivities such as peat makes accurate quantification of this component of the hydrologic budget difficult (Owen 1995).

Wetland ecosystems are particularly susceptible to changes in the timing and quantity of water they receive, as this affects both plant community structure and composition. Community changes occur primarily as a result of variation in flood tolerances among plants and the effects of flooding on growth rates (Ernst & Brooks 2003). Studies have shown that prolonged flooding causes a compositional shift toward more flood-tolerant tree species through the elimination of less flood-tolerant ones (Ernst & Brooks 2003).

Within the water column, changes in hydrologic conditions can either directly modify or alter chemical and physical properties such as nutrient and toxicant availability pH, salinity and dissolved oxygen concentrations, in addition to the degree of substratum anoxia, sediment geochemistry properties and interstitial fauna and flora (Sriyaraj & Shutes 2001).

Direct habitat alteration

Direct habitat destruction and alteration are two of the main causes of global coastal wetland decline. Urban centres have often developed in estuaries and today few of these remain unaffected by human activities (Lindegarth & Hiskin 2001; Heap *et al*. 2001). Upland deforestation due to urban development increases soil erosion within catchments and thus increases the sediment load entering receiving waterways leading to wetlands (Costanza & Greer 1998). Fragmentation of wetland habitats has been observed to have detrimental impacts on flora and fauna, causing changes in community composition and ecosystem function (Faulkner 2004). These impacts have been associated with changes in habitat patch size and thus changes in patch area–to–edge ratio, resulting in increased edge effects. For example, an increased area of forest edge to forest interior can create greater opportunity for invasive weed species to occupy an area of wetland habitat.

Fragmentation of wetland habitats can also impact on the fauna that depend on these ecosystems for habitat and food, particularly those with specific needs (Schiller & Horn 1997). Observed changes in community structure of shorebird assemblages in the northeastern United States were consistent with declines in forest interior relative to edge, caused by fragmentation from surrounding residential and urban development (Allen & O'Connor 2000). As a result of urbanisation, wetland areas are also affected at the ‘complex level’ through drainage modification and at the individual level through modification, isolation or fragmentation. Conversely, aquatic organisms may demonstrate different patterns of effect from wetland fragmentation, as many recent studies suggest that it is the interface between vegetated and unvegetated areas, that is, the ‘edge’ of coastal wetlands that provides the greatest attraction to organisms such as crustaceans and fish (e.g. Vance *et al.* 2002). Predation rates of shellfish in seagrass beds are dependent on the patch size–to–perimeter ratio (Irlandi *et al.* 1999).

Development of land close to existing wetlands often involves the disturbance of {potential?} acid sulfate soils (ASS) that either are or once were part of a wetland. After ASS have been oxidised, rainfall results in the flushing of acid from the soil. Runoff helps transport the acid to local waterways. This exposes marine organisms to rapid changes in pH, hypoxia, toxic levels of aluminium and manganese, iron precipitation and hydrogen sulfide. A recent study comparing the release of toxic metals from ASS with industrial effluents has found that such leacheates carry comparable concentrations of aluminium, cadmium, cobalt, manganese, nickel and zinc to the aquatic environment (Sundström *et al.* 2002). Major environmental impacts caused by ASS within waterways include fish kills and diseases, habitat degradation and changes to aquatic plant communities

(Cook *et al.* 2000). Recent estimates suggest that globally around 24 million hectares of coasts are affected by an acid sulfate problem (Ritsema *et al.* 2000), with large areas of a similar nature already developed for urbanisation or other human uses.

Conversion into ponds for finfish and prawn aquaculture, an indirect effect of urbanisation, has reduced coastal mangrove area by more than 50% in most southeast Asian countries (Wilkinson *et al.* 1994). Excavation of natural wetlands for conversion into aquaculture ponds often triggers acid sulfate problems. In developed countries such as Australia, wetland loss around urban centres is mainly through reclamation for residential use, such as in canal estates.

In southeast Queensland alone, it has been estimated that over 1200 hectares of mangroves and almost 600 hectares of saltmarsh-claypan areas were lost between 1974 and 1987 (Hyland & Butler 1988). During the same period, the area of artificial waterways and canals increased to about 5% of the area of natural mangroves and saltmarsh-claypans in the region. Artificial waterways created over natural wetlands can provide habitats for selected estuarine fauna, as surveys suggest that the canals support different fish assemblages compared to natural wetlands (Morton 1989, 1992), which may also have different trophodynamics (Connolly 2003).

More subtle effects of the indirect disturbances of urbanisation on wetland landscapes may also exist. For example, runnelling that involves constructing wide, shallow channels to increase tidal flushing of coastal wetlands, is a technique for effective mosquito control in urbanised soft-sediment coasts (Hulsman *et al.* 1989). Through modifying the hydrological regime, runnelling affects wetland community structure such as the burrow size and density of grapsid crabs (Breitfuss *et al.* 2004) as well as reducing mosquito populations. However, long-term research at the Coomera Island reference site in southeast Queensland has shown minimal effect generally on the saltmarsh. Over the first 14 years after the site was runnnelled in 1985, the vegetation processes were not significantly affected (Dale & Dale 2002). Although mangrove propagules are transported along runnels (Breitfuss *et al.* 2003) the general increase in mangroves in the area is not related to runnelling (Jones *et al.* 2004). With 19.5 years of post-runnelling data, the only effects are slightly increased soil moisture (because tides flood the area more often) and slightly less saline soil moisture near runnels (because of flushing). There were no effects on the soil or watertable pH or on watertable depth and salinity (Dale, 2005 unpublished data). Runnelling is unlikely to lead to acid sulfate problems as it increases wetness and seawater acts as a buffer to acidity (Saffigna & Dale 1999; Alsemgeest *et al.* 2005).

Nutrient loads

Urbanisation activities can impact on nutrient cycling primarily due to changes in hydrology and nutrient loadings. Such changes can alter plant species composition and nutrient cycling patterns. These impacts can, in turn, alter the species richness and abundance of bird, fish and macro-invertebrate populations (Faulkner 2004). The deforestation associated with urban development may also contribute to increased nutrient loads flowing into receiving wetlands as riparian vegetation provides a sink for nutrients and converts them into less harmful substances (Faulkner 2004). Similarly, alteration of the groundwater flow characteristics by urbanisation will probably also affect the nutrient dynamics of coastal wetlands. The role of subsurface flow on nutrient dynamics in coastal wetlands is, however, poorly known.

The impact of nutrient enrichment on coastal wetlands will depend strongly on vegetation type and background nutrient levels (Morris & Keough 2003). For example, mangroves and salt marshes are often nutrient-limited communities that would benefit from moderate nutrient enrichment (e.g. Feller *et al*. 2003), while seagrasses generally respond negatively to such enrichment (Hemminga & Duarte 2000).

Eutrophication and organic enrichment

Acceleration of eutrophication is one of the most significant anthropogenic processes in coastal waters (Rosenberg 1985; Nixon 1995). Increased human activity results in increased nutrient and organic matter discharge into coastal wetlands, whether directly from agricultural runoff or indirectly through discharges such as treated effluents. Given the historical trend of human settlement concentrating around estuarine wetlands, anthropogenic-sourced nutrients and organic matter have become an increasingly important component of the material budget of urbanised coasts. This is especially true of coastal wetlands, which typically occur on low-energy, depositing shores. Li and Lee (1998) estimated that anthropogenic sources accounted for roughly half of all the available organic carbon on which the fishery and waterfowl species in Deep Bay (an urbanised embayment in Hong Kong, southern China) depend.

Urbanisation exerts a demand for services such as wastewater treatment (Faulkner 2004). Treated sewage effluents typically contain toxic metals and are high in nutrients, particularly nitrogen and phosphorus. Bioactive substances such as drug residues and endocrine-disrupting substances are also major concerns (Depledge & Billinghurst 1999). Such nutrient fluxes often cause shifts in phytoplankton community and composition culminating in blooms (e.g. Bowen &

Valiela 2001). They also result in eutrophication when the algal bloom dies and microbial bacteria exert an increased oxygen demand on the ecosystem during decomposition of the dead algae (Connell *et al.* 1999; Rapport *et al.* 1998). Similar blooms in micro-algae have been observed to increase turbidity and contribute to the loss of benthic macrophytes. For example, Costanza and Greer (1998) reported loss of seagrass in Chesapeake Bay estuary preceding phytoplankton blooms. This reduction in benthic plant cover destabilises the benthos and provides a positive feedback loop for sedimentation and turbidity within the system (Costanza & Greer 1998). The effect of increased nutrient loading is exacerbated by modifications to wetland hydrology associated with urbanisation of the surrounding environment through dyking and other forms of impoundment to result in restricted tidal flows (Fong & Zedler 2000).

While the effect of eutrophication and organic enrichment on macrobenthic structure may be predictable (Pearson & Rosenberg 1978), the implications of such changes for the beneficial ecosystem services offered by coastal wetlands can be subtle. Lee (2003) reported that the number of waterfowl overwintering in Deep Bay, a eutrophic, mangrove-fringed wetland in south China, was correlated positively with water biochemical oxygen demand and total nitrogen load. This finding supports the trophic analysis of the same wetland using both the mass balance and stable isotope approaches (Li & Lee 1998; Lee 2000), in that food chains leading to the waterfowl populations are dependent on anthropogenic rather than natural sources in eutrophic environments. Simple degradable organic matter and moderate levels of nutrient enrichment would have an initial beneficial effect on all but the most nutrient-rich communities, but the positive impacts have to be balanced against the risk of the system ‘collapsing’ upon reaching beyond the ‘ecotone’ point (Pearson & Rosenberg 1978).

Toxic pollutants

Pollutants reach wetlands primarily through catchment and stormwater runoff. Urbanised catchments collect large amounts of pollutants, including eroded sediments from construction sites, trace metals and petroleum wastes from roadways and industrial and commercial areas, and nutrients and bacteria from residential areas (Horner 2000). Just as urbanisation produces larger quantities of pollutants, it reduces water infiltration capacity, yielding situations more vulnerable to concentrated transport by surface runoff than pollutants from other land uses.

Within the complex web of wetland sediment–water components, the movement, availability and possible toxicity of contaminants are affected by chemical and physical factors that are in turn influenced by hydrological parameters that include residence time, tidal flushing, reduction-oxidation (redox), pH and salinity gradients and temperature (Lau & Chu 1999). Seasonal changes in these factors further compound their importance in governing the levels of bioavailable nutrients within the system (Lau & Chu 1999). Coastal wetlands that are subjected to periodic flooding may demonstrate large temporal variability in these and other hydrological parameters as sediment–water nutrient exchange is usually heightened during peak flow periods. Modification of the runoff pattern by urbanisation will likely have the same effect in causing larger variability in nutrient exchanges than that of natural wetlands.

The efficacy of wetlands in removing pollutants from the upslope surface water and groundwater is highly dependent upon hydrology (Corbitt & Bowen 1994). In freshwater wetlands, factors such as the frequency and timing of sampling (e.g. high or low flow period) may affect their apparent effect on nutrient concentrations (Fisher & Acreman 2005). For effective removal of pollutants, sheet rather than highly focused flows must occur and advance at a slow velocity and shallow enough depth to allow interaction with the sediment–water interface (Prior & Johnes 2002). Mechanisms responsible for retention of non-toxic contaminants such as nitrogen (N), phosphorus (P) and suspended sediment include denitrification and assimilation of N, precipitation and sorption of P, trapping of sediments and adhering P and uptake by vegetation. The nutrient removal capacity of a wetland decreases with the flow of water through the riparian zone. In addition, high flows may lead to net nutrient release through the flushing of nutrient-rich soil-water, desorption processes and sediment erosion (Prior & Johnes 2002). Longer residence times experienced during the growth season, however, promote removal processes, particularly denitrification (Prior & Johnes 2002), effectively increasing nutrient retention. For this reason, subsurface flows in wetland ecosystems are often associated with higher rates of removal than surface flows. Upward trends in nutrient enrichment of surface waters resulting from diffuse sources have produced concerns about their ecological impacts on wetland systems (Prior & Johnes 2002).

Many toxic inorganic and organic pollutants are of great concern to water quality managers owing to their persistence, toxicity and likelihood to bioaccumulate (Turner & Tyler 1997). The major temporary or ultimate sink for such pollutants in coastal wetlands and estuaries is the sedimentary reservoir. Therefore, the definition of the biogeochemical mechanisms by which they adsorb onto, desorb from and repartition amongst natural, heterogeneous particle populations is essential in order to assess their environmental fate (Dyer 1995; Turner & Tyler 1997; Whitehouse *et al*. 2000). Within intertidal urban wetlands, the prediction of pollutant distribution is further compounded by intense temporal and spatial gradients of reaction controlling variables such as salinity, dissolved oxygen concentration and particle composition, occurring both within the sediment and in the water column during particle suspension (Turner & Tyler 1997).

At the same time that urbanisation produces larger quantities of pollutants, it reduces water infiltration capacities, making water more vulnerable to transport by surface runoff than pollutants from other land uses. Increased surface runoff combined with disturbed soils can accelerate the entrainment of sediments and the transport and deposition of sediments from the catchment into downstream coastal waters and wetlands (Ehrenfeld 2000; Horner 2000).

An increased use of pesticides, heavy metals, cleaning agents and petroleum products occurs in urbanised environments (Costanza & Greer 1998). These chemicals accumulate in the catchment and are either washed off the land or through stormwater drains during periods of heavy rainfall into creeks and rivers that flow into wetlands and estuaries (Costanza & Greer 1998), degrading water quality and habitat (Faulkner 2004). With increases in the proportion of impervious surfaces resulting from urbanisation, the impact of more concentrated stormwater runoff in the translocation of chemical pollutants to coastal wetlands is a growing concern. A portion of these pollutants can accumulate in the tissues of wetland organisms and biomagnify through the food web (Connell *et al*. 1999).

Physical, chemical and biological processes interact to manipulate the retention, transformation and release of a large variety of sediments and chemical species within intertidal water bodies and their associated behaviours. Additionally, increased peak flows as a result of increased impervious surfaces through urban development transport more sediment to wetlands, which may alter vegetation community structure, impacting on both benthic and other animal species dependent on it (Nakamura *et al*. 1997).

Changes in hydrology can also affect nutrient transformations, availability and deposition and the flux of organic materials within a system (Whitehouse *et al*. 2000). Greater surface runoff is also likely to increase the velocity of inflow to wetlands, which can potentially disturb resident biota and scour substrata. The increased runoff impacts are greater on maintaining the values and services of the remnant terrestrial vegetation, resulting in loss of leaf litter and surface soil organic matter from these areas and further adding to instream loads. Sediment retention within coastal wetlands and estuaries is directly related to flow characteristics, including degree and pattern of channelisation, flow velocities and storm surges (Ehrenfeld 2000; Horner 2000).

Sedimentation

Hydrological characteristics within wetlands directly influence the rate and degree of sedimentation of solids imported by runoff into a receiving system. Excessive sedimentation as a result of urbanised catchments may alter wetland topography and soils and ultimately result in infilling (Horner 2000). Alternatively, elevated flow velocities can scour a wetland's substratum, changing soil composition and leading to a more channelised flow. This can itself then result in greater velocities, increased erosion and a greater concentration of suspended sediment load within the water column (Whitehouse *et al*. 2000).

Deposition of increased sediment loads within estuaries can reduce flow through wetland systems, providing a positive feedback loop for further increases in sediment deposition within the system (Callaway & Zedler 2004). Such alterations in flow have been associated with changes in water salinities, temperatures and oxygen concentration and in extreme cases may contribute to eutrophic conditions (Fong & Zedler 2000) and subsequent fish and invertebrate kills (Callaway & Zedler 2004).

Vegetation plays an important role in the shaping of depositional forms within intertidal wetlands and on shores experiencing estuarine conditions. Algae and seagrasses partially stabilise the surface of tidal flats, and salt-tolerant plants such as mangroves colonise the estuary margins spread across the intertidal zone (Bird 1984).

Colonisation by plants such as reeds and mangroves can occur quickly on newly deposited sediments, if permitted by the prevailing hydrological and wave exposure regimes. Such colonisation is likely to interact synergistically with reduced flows and increased sediment deposition, providing an additional feedback loop for the deposition of yet higher sediment loads in urban-impacted wetland systems. Rapid changes in coastal wetland vegetation pattern have been documented to result from wetland landscapes experiencing significant anthropogenic modifications (e.g. Lee 1990). In subtidal areas, increased sedimentation also enhances turbidity and can stress benthic macrophytes such as seagrass (Hemminga & Duarte 2000). Alteration in sedimentation regime may have contributed to the recently documented phenomenon of mangrove intrusion into salt marshes in Australia, reportedly caused by increased nutrient and freshwater delivery (Saintilan & Wilton 2001).

While within-habitat macrofaunal structure may not be correlated strongly with small-scale changes in sediment characteristics (Chapman & Tolhurst 2004), changes in vegetation cover or sediment composition often trigger shifts in macrofaunal assemblages (Lui *et al*. 2002; Thrush *et al*. 2003), which are highly responsive to changes in organic matter content, water depth and other sediment characteristics such as pH and redox regimes. Sedimentation from urbanisation *per se* also results in changes to macrofaunal assemblage structure and abundance, as chemicals carried by the sediment along developed coasts have strong effects upon them (Inglis & Kross 2000; Ellis *et al*. 2004).

Sedimentation in ponds and wetlands is important for removing not only the sediment itself but also the nutrients and contaminants that readily attach to fine particles (Walker 2001). Suspended matter has a strong tendency to absorb pollutants. Sediments are major contributors in the removal (or sink) of pollutant in wetlands (Turner & Tyler 1997; Whitehouse *et al*. 2000). There have been a number of sedimentation studies undertaken in wetlands, both in situations where there was an essentially constant flow-through and where the main flows were due to runoff from storms. In the latter case, the intermittent nature of stormwater wetlands leads to a situation where sediment is more likely to be distributed around the entire basin (Walker 2001).

In order to understand how sedimentation affects the long-term sustainability of coastal wetlands such as mangroves, it is important to understand the sediment dynamics of such systems. The processes controlling sediment dynamics in vegetated wetlands are, however, not fully understood. It is therefore often difficult to define the sediment budget of an individual degraded wetland and the role the budget plays within the dynamics of the larger coastal sedimentary system (Kitheka *et al*. 2003).

In lowland anastomosing wetland systems, sediment deposition from overbank flow is a critical component of lateral connectivity between river channels and their floodplains. This connectivity sustains riparian ecology and biodiversity, which may significantly reduce a wetland river system’s total suspended sediment load (Wolanski 1992). Understanding the relationship between floodplain sediment assemblages, geomorphic processes and land uses is significant for predicting changes in depositional processes that result from initial anthropogenic disturbances and later attempts to rehabilitate habitats in lowland wetland systems.

Urbanisation effects on sedimentation

Urbanisation generally increases the amount of sediment carried by terrigenous water supply to coastal wetlands. Sediments within the system can remain either in suspension or on the substratum until it is disturbed from a number of sources, including natural causes such as aquatic fauna activity or resuspension due to tidal flow. Human activity such as boat movements can also contribute to the mechanisms causing resuspension.

The supply of sediments generated by urbanisation is usually within the range that can be tolerated by mangrove wetlands and in moderate volumes is usually essential in substratum accretion which helps plants keep pace with sea level rise (Wolanski 1992). However, during periods of high sediment supply, particularly those associated with extremely high river discharges, enormous volumes of terrigenous sediment are usually discharged into mangrove wetlands. This also occurs as a consequence of natural flood events.

Sediment contamination not only threatens biological life through bioconcentration (bioaccumulation from water) and biomagnification (bioaccumulation from food), but also affects the ecosystem contaminant dynamics (Lau & Chu 2000). Additionally, heavy siltation raises the elevation of the wetland so that inundation during flood tide is restricted to zones fringing tidal inlets and main channels (Kitheka *et al*. 2003).

Water quality impacts on coastal wetland sediments can eventually threaten the existence of a wetland. Where sediment inputs exceed rates of sediment export and consolidation, a wetland may gradually become filled. Filling by sediment is a particular concern for wetlands in urbanised areas, as many of them have an ability to retain great volumes of sediments (Horner 2000). Enhanced sedimentation results in rapid changes in the form and function of many systems. A key issue is the invasion of terrestrial or semi-aquatic vegetation in highly sedimented areas. These alter flows, trap nutrients and cause water quality issues when plant material decomposes.

Within urbanised and degraded catchments, catastrophic siltation events have been recorded following extreme rainfall events (Brooke 2002) . During such events, large ‘slugs’ of sediment from urbanised catchments move downstream into coastal water bodies, rapidly infilling channels with coarser material while finer sediment is deposited across much of the system. The finer-grained sediments that were originally deposited in shallow areas are continually remobilised by wind-generated waves, producing chronic turbidity (Brooke 2002) and changes to tidal and current regimes within intertidal wetlands (Bird 1984).

Fine sediments derived from the catchment and produced within the estuary by the decomposition of biota may also flocculate and settle in the margins of the estuary, forming mudflats where there may have formerly been relatively clean sand (Brooke 2002). Additionally, in association with increased rates of sedimentation, the amount of sediment-bound nutrients—such as total phosphorous, total nitrogen or total carbon—entering estuaries from their catchments may also increase as a result of urban expansion. As a consequence of increased nutrient inputs, infilling is enhanced, even where the volume of terrestrial sediment influx is low, due to the increased amount of organic material accumulating in the estuary. In combination with high turbidity, these pressures can lead to the loss of healthy benthic habitats.

High suspended material inputs can reduce light penetration, dissolved oxygen levels and overall wetland productivity (Horner 2000). Pollutants bound to sediments in runoff (Turner & Tyler 1997; Whitehouse *et al*. 2000) can interfere with the biological processes of the aquatic flora and fauna, resulting in impaired growth, mortality and changes in community structure (Horner 2000)**.** Ellis *et al.* (2004) documented significant shifts in mangrove health and macrobenthic community structure in response to sedimentation composition driven by catchment land-use pattern in two estuaries in New Zealand.

Topography and hydrological changes may effect the stability of the benthic substratum through alteration in water current speeds, increasing turbidity and the potential for increased rate of pollutant transportation within the water body as well as indirectly and directly affecting resident organisms (Horner 2000). Additionally, an increase in the proportion of impervious (sealed) surfaces within the catchment and the presence of efficient drainage systems has altered and often increased the flows and pollutant loads carried by stormwater to local waterways (Bingham 1994; Horner 2000).

In coastal marine environments, the chemical transformations that take place at the sediment–water interface determine the cycling of nutrients and pollutants between the sediment substratum and the overlying water column (Spagnoli & Bergamini 1997; Whitehouse *et al*. 2000). The former can constitute either a source of or a sink for nutrients, so that in shallow areas sediments can be one of the major factors that control the trophic level of the aquatic system (Spagnoli & Bergamini 1997). Many of these chemical reactions are biologically mediated; their relative importance depends on several factors such as sediment composition, sedimentation rate, hydrodynamics, bioturbation and irrigation, as well as the physical and chemical characteristics of bottom waters (Spagnoli & Bergamini 1997; Francois *et al*. 2002; Nickell *et al*. 2003).

The sediment characteristics within the system have many implications for the health of the overlying water body. When this sediment is stirred up, trace metals, nutrients and organic contaminants are released into the water column, limiting the amount of light essential for plant growth (Dyer 1995). Contaminants such as heavy metals, nutrients, micro-organics, pesticides and herbicides have a strong tendency to adsorb to fine-grained sediments, thus relating pollutant transport and storage strongly to sediment dynamics.

An understanding of the sediment sources delivered to, stored within and exported from intertidal waters and wetlands is important for a number of environmental issues including maintenance of navigational channels, light availability for primary productivity, reduction of dissolved oxygen concentrations and the transport and accumulation of particle-bound nutrients and contaminants and their eventual transport to the continental shelf (Eyre *et al*. 1998). Most of our understanding of the relative contributions from the various sediment sources, fluxes and storage in estuaries is derived from studies of large, temperate, northern hemisphere estuaries (Eyre *et al*. 1998).

Sediments within the system can remain in suspension or on the substratum of the water body until they are disturbed by a number of sources, including natural causes such as aquatic fauna activity or resuspension due to tidal flow. Human activity such as boat movements can also contribute to the mechanisms causing resuspension. Once initially disturbed, sediment is more likely to be resuspended into the water column because of the unstable bed conditions produced following the settling period. The major biological and physical disturbances that impact on intertidal sediments include tides, waves, storms, runoff events, macrofaunal activities (bioturbation or bioirrigation) and anthropogenic factors such as fishing and dredging (Dyer 1995, 1997; Whitehouse *et al*. 2000; Webb & Eyre 2004). The disturbances occur over a wide range of time scales and influence processes over different space scales (Table 1-2). Disturbances have a net effect in producing environments that, although broadly structured, are patchy and dynamic.

Resuspension processes in coastal systems affect the cycling of sediments, nutrients, carbon and contaminants. The disturbances are ubiquitous processes influenced by a combination of the strength, speed, direction and duration of both winds and currents respectively, and by topographical characteristics (Shteinman *et al*. 1997). This is especially true in shallow water bodies.

Table 1-2. Spatial and temporal scales of disturbance of the intertidal environment (modified from Turner & Tyler 1997)

Disturbance	Spatial scale [a]	Temporal scale [b]
Irrigation	µm – mm	s – h
Biogeochemistry	µm – mm	s – h
Bioturbation	mm – m	s – d
Waves	m	s – h
Fishing	m	s – h
Dredging	m – km	h – d
Runoff events	m – km	h – d
Wind/storms	m – km	s – d
Tides	m – km	h – d
Sedimentation/erosion	µm – mm	h – dc
Introduction of exotic species	m – km	y – dc
Eutrophication	m – km	y – dc
Global change/sea level rise/ geomorphology	µm – m	y – dc

[a] µm = micrometres, mm = millimetres, m = metres and km = kilometres
[b] s = seconds, h = hours, d = days, y = years and dc = decades

Impacts of urbanisation: a summary

In addition to causing direct habitat loss, urbanisation impacts on the structure and function of coastal wetlands through its effect upon the hydrological and sedimentation regimes and the dynamics of nutrients and pollutants—the major drivers of wetland dynamics. The ecosystem services offered by urbanised wetlands are then compromised by secondary changes in species composition and dominance, habitat connectivity, productivity and metabolism. There is, however, scant unequivocal evidence to support these impacts, mainly because of a lack of knowledge about responses at higher levels of organisation and because of methodological issues such as those described below.

Methodological issues associated with studying urbanisation impacts on coastal wetlands

Odum (1985) first described how ecosystems might respond to natural and anthropogenic stresses, hypothesising changes in the energetics, nutrient cycling, community structure and other aspects of ecosystem structure and function. Such ideas have since been further developed to generate the paradigm of ‘ecosystem health’ (Rapport *et al*. 1998). However, assessment of disturbance effects on coastal wetlands at the ecosystem level has been scarce.

Hopkinson and Vallino (1995) reviewed the impact of human activities on runoff patterns, and how these might trigger changes in estuarine community metabolism. Rapport *et al.'s* (1998) idea of the 'ecosystem distress syndrome' is based on the theory that ecosystem health as a whole declines in response to anthropogenic activities. Ecosystem indicators such as changes in primary productivity and nutrient sources, reduced species diversity, horizontal nutrient transport, prevalence of disease and parasitism, extinction of habitat specialists and reduced mutualistic interactions between species are measured to assess the health of impacted ecosystems (Rapport 1998).

These predictions have, however, seldom been tested. This is not surprising, as even data on ecosystem dynamics of relatively undisturbed systems are rare. Hopkinson (1992) presents one of a few tests of the predictions on ecosystem development pattern by Odum (1969) on undisturbed systems, comparing the effects of openness and vegetation (forested versus marsh) on the metabolism of four freshwater wetlands. Probably due to the high complexity typical of ecosystem level measurements, the study by Hopkinson (1992) was limited in replication (one site only for each combination of openness and vegetation type) and thus in generalising ability. The lack of such 'baseline' data on wetland ecosystem dynamics makes the detection and prediction of stress effects highly difficult.

Figure 1-1 provides a theoretical framework against which the impact of urbanisation on coastal wetland structure and function may be tested. Various drivers, processes and feedback loops have been identified that eventually lead to indicators of ecosystem stress such as changes in primary productivity sources and level and species diversity. In addition to the need to construct functional models for coastal wetlands so as to allow hypothesis testing in relation to urbanisation impacts, there is also the need to improve the design of such ecosystem level studies. Most past reports on the impact of urbanisation address responses of coastal wetlands in a piecemeal manner, with little control and replication in the comparisons that form the basis for 'impact'. Since most studies on the impact of urbanisation are 'natural experiments' largely out of the control of the researcher, temporal replication is usually difficult to achieve. Spatial replication, however, is often possible.

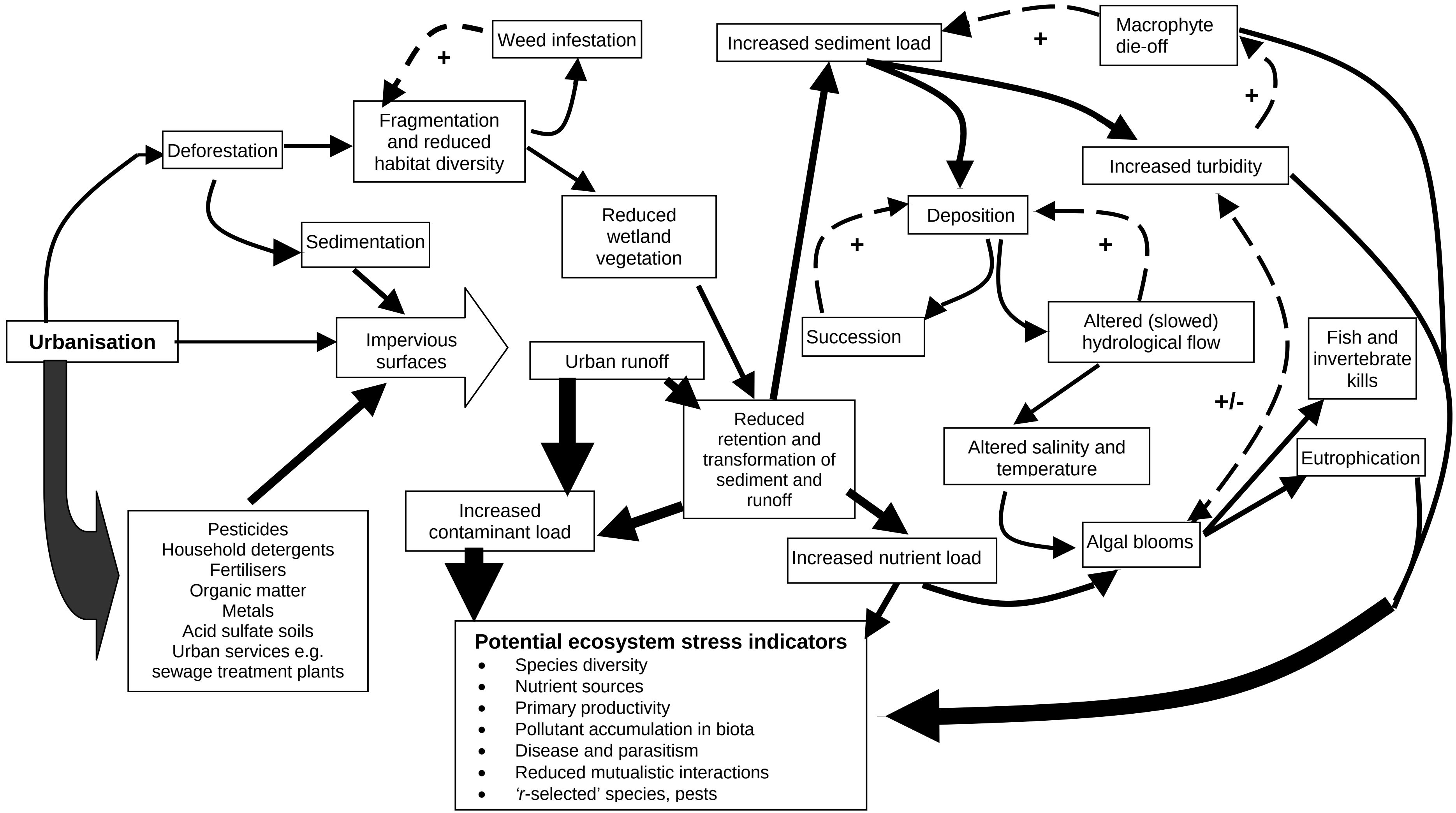


Figure 1-1. A summary of the impacts of urbanisation on wetland ecosystem function showing disturbances, processes and feedback loops

The strength of the effects and interactions are indicated approximately by the width of lines joining individual components. Feedback relationships are indicated by dotted lines, with direction indicated by (+/-). Reproduced from Lee *et al.* (2006).

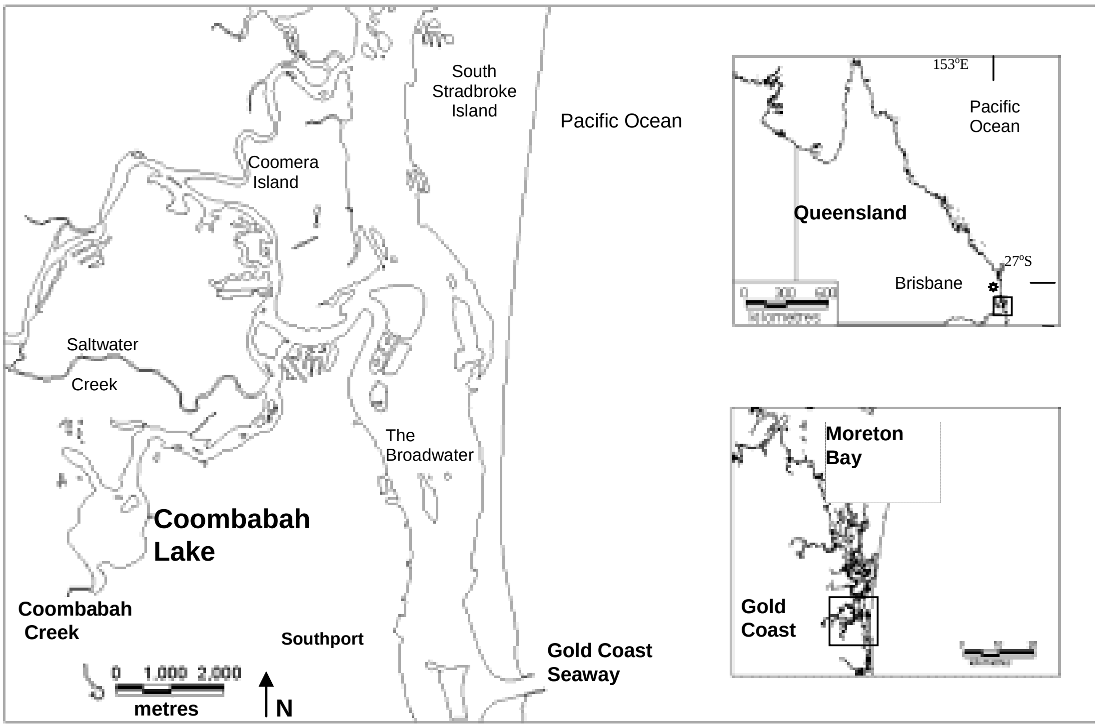
Studies that are not spatially replicated will not allow unequivocal detection of impact (Green 1979), especially more so when the anticipated variability in the comparison criteria is high, as is expected of highly dynamic systems such as coastal wetland ecosystems. This high variability also requires that multiple reference locations be used (an asymmetric impact study or 'beyond-BACI' design, Underwood 1994) to ensure that no erroneous conclusions are reached simply because of the choice of a single reference location. In a study on the impact of boardwalk construction on mangrove macrofauna, Kelaher *et al*. (1998) demonstrated large differences in response among study locations to the same disturbance, again pointing to the need for spatial replication. Ecosystem level indicators of urbanisation effects (e.g. community metabolism pattern) are expected to be more variable than lower level indicators (e.g. sedimentation rate) (Connell *et al*. 1999), thus requiring even more extensive replication to confirm impact.

An alternative to the beyond-BACI approach is the use of multiple study locations along a defined urbanisation gradient to formulate trends that relate ecosystem level indicators to the degree of urbanisation. The application of this approach however requires an acceptable metric (e.g. ratio of impervious to pervious surfaces in a catchment) to be devised as an indicator of the level of urbanisation. Given the many immediate effects of urbanisation on natural ecosystems, the metric would need to incorporate and represent the impacts in a meaningful and unbiased way. This approach has the advantage of not requiring study locations to be categorised rigidly into 'impacted' or 'unimpacted' groups, as today almost no location can really be regarded to be free from human impact.

Further, there is often the practical difficulty in delineating catchments or sub-catchments in assessing the extent and impact of urbanisation. Traditional elevation models based on broad contour information with typical resolution at around half a metre are inadequate for identifying hydrologic connectivity through tidal flushing or terrestrial runoff. Digital elevation models constructed using aerial laser surveys offer significantly improved resolution but such information is still far from being widely available. As habitat connectivity through water flow has strong implications for material transport, movement of the biota and general coastal wetland function, the lack of such critical information presents a major barrier to the assessment of the impact of urbanisation.

Coombabah Lake case study

This study investigates the impact of urbanisation on coastal wetlands through a detailed case study of Coombabah Lake (23°55' S 153°21' E) in southeast Queensland (see Figures 1-2 and 1-3). Coombabah Lake has an area of about 460 hectares and is hydrologically linked to the tidal broadwater and the upstream catchment in the suburbs of Gaven and Ernest via Coombabah Creek. The lake is surrounded by tidal wetlands dominated by mangroves (mainly *Avicennia marina* and *Aegiceras corniculatum*, with some occurrence of *Rhizophora stylosa*) and salt marshes (mainly *Sporobolus virginicus*), which are fringed on the seaward side by tidal mudflat and on the landward side by melaleuca forests.



In view of its value as an important habitat for juvenile fish, Coombabah Lake was afforded protection under Queensland Fisheries legislation as "a prohibited area for the taking of all or any kind of fish" in 1952. This prohibition of fishing within the lake remains current under the present fisheries legislation and in 1983 was complemented by declaration of the lake as a wetland reserve (termed Level B fish habitat area under current fisheries legislation). Level B fish habitat area status provides increased management of all subtidal and intertidal habitats within the lake and protects these habitats from major, direct, development impacts. This protection did not extend to the supralittoral habitats, however, which have increasingly been turned into urban developments, especially on the northwestern shores of the lake. The Helensvale development was established in the 1970s, and large areas of the upstream catchment in the suburbs of Gaven and Ernest have been developed for urban and industrial purposes.

Figure 1-3. An aerial view of Coombabah Lake and its associated habitats from the south

Helensvale development, the largest urban area around the lake, is in the far background. Large urban developments are visible in the top left-hand corner as white areas. The sewage treatment plant is near the top left, with the 'STP creek' draining into Coombabah Lake as indicated by the white arrow. The occurrence of mangroves and melaleuca forests along the tidal gradient can be seen in the foreground. (Photo courtesy of Matt Leon, Griffith University, Gold Coast campus)

Coombabah Lake embodies the type of urbanisation pressure experienced by wetlands in fast developing coastal areas. With an additional one million people expected to migrate to the coastal region of southeast Queensland in the next 25 years, increasing urbanisation is inevitable and will affect the capacity for the surrounding coastal wetlands to sustain their beneficial ecosystem services.

The current study incorporates sediment, water, nutrient and fish components, and aims to characterise the present condition of the lake in terms of water exchange, circulation, sediment and pollutant resuspension and accumulation. It also assesses the impact of local urban runoff on ecosystem structure and function. While it is desirable to conduct assessments at the lake level, for example, by across-lake comparisons, this study was restricted in scope—attempting to detect impact at local 'hotspots' of urban influx within Coombabah Lake—because of limitations in time and funding. Detailed information on methods is given in each of the technical reports (see Chapters 2–5).

Need for this study

Coastal wetlands are globally under threat from urbanisation. In Australia, due to the concentration of the population within a short distance from the coastline, the pressure for developing within the low-lying and flat coastal wetlands is paramount. Large areas of mangroves and salt marshes have been destroyed in southeast Queensland by urban development in the 1970s (Hyland and Butler 1988). However, little is known about how urbanisation may affect the capacity of coastal wetlands to sustain beneficial ecosystem services. Even less information is available on actual measurements of impact. Such information is critical for the planning and management of wetlands resources on rapidly urbanising coastlines.

This research aims to help fill these gaps in knowledge through a detailed case study of Coombabah Lake, a coastal wetland with recognised ecological significance in rapidly urbanising southeast Queensland.

Objectives

The objectives of the study are to:

1. Characterise the present sediment condition of Coombabah Lake using a suite of physical, chemical and biological parameters
2. Investigate the hydrology and sediment transportation and accumulation in the lake
3. Assess the impact of urban influx on Coombabah Lake with selected biotic indicators.

Approach and methods

The study provides a detailed snapshot of an urbanising coastal wetland, as an approach to understanding how urbanisation may affect the condition of coastal wetlands. While wetland-level comparisons are capable of providing a more broadly applicable evaluation of urbanisation impacts than within-wetland studies, the former approach demands time and effort beyond the resources of the present study. Wetland-level comparisons also are often constrained by strong differences in local conditions such as geology, tidal regimes and management practices (e.g. control of fishing activities).

The Coombabah Lake study was designed following the conceptual framework of urbanisation effects depicted in Figure 1-1. Not all the processes or aspects of urbanisation influence are studied; rather, particular attention was paid to the physical elements of water and sediment as these are the major determinants of wetland structure and function. The study then investigated the effects of urbanisation by assessing the condition of the ecosystem through a number of biotic indicators.

Presentation of research findings

The technical reports presented in Chapters 2–5 each describe a specific set of research activities undertaken as part of the case study of Coombabah Lake:

- Characterisation of the physical, chemical and biological composition of lake sediments (Chapter 2)
- Components of sediment transport in the lake (Chapter 3)
- Patterns of tidal flushing at the mangrove fringe of the lake (Chapter 4)
- Detection of urban flux effects in the lake using biotic indicators (Chapter 5).

Each chapter follows the same format and contains:

1. Chapter summary – a synopsis of the aims, investigations and findings of the work
2. Background – information providing context and relevance
3. Approach and methods – detailed description of the research carried out
4. Results and discussion – analysis of findings
5. Conclusion – final observations on the study's outcomes and, where relevant, any further development that would be beneficial.

Chapter 2

Characterisation of the physical, chemical and biological composition of lake sediments

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Chapter summary

These investigations aimed to determine (i) the composition and behaviour of the sediments in Coombabah Lake; (ii) if and how urbanisation has affected the rate and nature of sediment accumulation, through a better understanding of current and former sedimentation conditions; and (iii) if there are any environmental implications for the health of the lake. Other specific objectives were to investigate the nature of nutrient and sulfur cycling associated with sediments, to scope the risk from a wide range of heavy metals in the sediments and to examine lake sediments in relation to the capacity to support fisheries resources.

The components of the research included:

1. A lake-wide survey of the properties of surficial lake sediments
2. A scoping study to determine overall metal and sulfide risk
3. A case study of monosulfide risk
4. An integrated iron, sulfide and metal case study involving a detailed risk assessment using geochemical interaction factors
5. An assessment of lake sediment history and urbanisation impacts using radiocarbon dating.

Increased concentrations for all nutrients analysed (including organic matter) were observed in samples dominated by high mud contents, occurring predominantly in the southern region of the lake. Average concentrations in sediments from the southern samples were greater for all nutrients measured compared to sediments from the northern, sandier half of the lake, where sediments originate from the coastal zone.

The present data on nutrient concentrations in surface sediments within Coombabah Lake indicates that the local environment is typical of Australian estuarine waters and is currently not heavily affected by urban expansion. However, we have no pre-urban baseline data to compare with. The observed urban-based influences entering the lake suggest that regular monitoring of sediment nutrient loadings be undertaken.

Enriched nitrogen stable isotope ($\delta^{15}N$) values were found predominantly at the entrances of the lake at both the marine and freshwater ends. Measured carbon stable isotope ratios ($\delta^{13}C$ values) were similar and within the range reported for mangroves. Additionally $\delta^{15}N$ values corresponded to previous studies of values approximating and below +2 ‰ for mangroves; however a large proportion of the sampled sediments indicate enriched $\delta^{15}N$ values (> +2 ‰).

Sediment quality assessment for metals and sulfides in Coombabah Lake was also undertaken. Sulfides were examined because they can potentially impact on dissolved oxygen levels in the lake and also can help buffer the bioavailability of sediment metals.

A scoping study across the lake found levels for heavy metals were generally below the ANZECC/ARMCANZ (2000) guidelines. Exceptions with nickel and arsenic (slightly above trigger levels) were found at the upstream reach and entrance of Coombabah Creek. Interestingly, samples in the vicinity of the sewage treatment plant and possibly acidic drains did not show elevated metal concentrations.

The levels of chromium reducible sulfur recorded in sediments are sufficient to trigger further acid sulfate soil investigation and management action should the sediments be disturbed for urban development or dredging. The level of monosulfides measured in these sediments is notably low compared to values recorded elsewhere, in New South Wales and Queensland estuaries. It was concluded that there was no evidence of significant risk to the level of dissolved oxygen in lake waters from the resuspension of sediments.

Results suggest that the concentrations of the metals tested (except possibly manganese) may be contained within the fine sediment fractions. The trends of increasing metal concentrations can be largely explained by natural gradients in the abundance of aluminosilicate minerals from the fine-grained sediments. Every assessment confirmed that overall the bioavailability risk from metals within Coombabah Lake sediments is currently low. However, the sediments have low and generally transient concentrations of acid volatile sulfides. This means that the lake sediments may not be able to tolerate significant additional metal sources and hence may not buffer the system sufficiently to prevent metals bioaccumulating indefinitely.

Using radio carbon dating, a shell layer found at 0.3–0.5 m below the surface is in the order of 2000 years old, suggesting that sediment accumulation in the lake is extremely slow and has not been greatly accelerated by urbanisation. However, the northern end of the lake does appear to be comprised of tidal delta sands that are actively accreting. The increasing level of tidal exchange now being experienced by the lake may act as a buffer, maintaining a well flushed and healthy sediment and related water quality environment.

Background

This chapter reports on selected characteristics and behaviour of the Coombabah Lake sediments. The lake is shallow, with a tidal range of less than 1 m, water depths ranging from 0–1 m in channels at low tide, and with large areas of sediment exposed at low tide. The nature and behaviour of sediments in such systems may impact on lake health and biodiversity by influencing turbidity, nutrient cycling and redox conditions of the water column and allow contaminants to enter the food chain.

The sediment transport systems of Coombabah Lake and its catchment play a critical role in the water properties and water quality of the lake. Sediment enters the system through a variety of sources, including urban runoff from within the catchment with rainfall, bank erosion and advection with tidal currents (Dyer 1995, 1997; Whitehouse *et al*. 2000). Potential sources of sediments, metals and nutrients to the lake are runoff loads from Coombabah Creek, tidal loads from the downstream system, diffuse sources from the adjacent urban development and, for nutrients and metals only, point sources associated with sewage pipes, the sewage treatment plant (STP) or the Suntown landfill. There could also be some atmospheric deposition of nutrients. Once the sediment is in Coombabah Lake, it can be transported out with the tide while suspended, or it may settle somewhere as bottom sediments.

The suspension, settlement and resuspension (of suspended sediment load) can affect the physical condition of the Coombabah Lake estuary system and the associated aquatic organisms within it. Suspended sediment limits the amount of light penetrating the water column (Dyer 1995), which may cause negative effects on living organisms that rely on photosynthesis for survival such as phytoplankton, zooplankton and possible seagrass bed colonies, and may also reduce the overall primary productivity of the lake.

Sediments provide an integrated picture of organic input and storage within an environment over significant time periods. Commonly used solid phase nutrients including total organic carbon (TOC), total nitrogen (TN), and total phosphorous (TP) concentrations may be used to investigate risks to water quality. They can be used to quantify the organic richness of sediments, with high values often found in environments characterised by high productivity, little oxidation of organic matter by aerobic processes and rapid burial and preservation of organic matter (Gallagher 2001). Under certain conditions, which create resuspension, bottom sediments may be released back into the water column, carrying pollutants and cycling them back into the water column (Dyer 1997; Mudroch & MacKnight 2000; Nickell *et al*. 2003). A particular distribution equilibrium of contaminants is established among suspended and bottom sediments, sediment pore water, overlying water column and biota. This equilibrium is affected by the physiochemical regime, which controls the kinetics of different reactions taking place within the system (Mudroch & MacKnight 2000).

In coastal marine environments the chemical transformations that take place at the sediment–water interface determine the cycling of nutrients and pollutants between the sediment substratum and the overlying water column (Spagnoli & Bergamini 1997; Whitehouse *et al*. 2000). The former can constitute a source or a sink for nutrients so that, in areas with shallow water, sediments can be one of the major factors that control the trophic level of the aquatic system (Spagnoli & Bergamini 1997). Many of these chemical reactions are biologically mediated; their relative importance depends on several factors such as sediment composition, sedimentation rate, hydrodynamics, bioturbation and irrigation, as well as the physical and chemical characteristics of the bottom waters (Spagnoli & Bergamini 1997; Francois *et al*. 2002; Nickell *et al*. 2003). Resuspension processes in coastal systems affect the cycling of sediments, nutrients, carbon and contaminants. Such disturbances are ubiquitous processes influenced by a combination of the strength, speed, direction and duration of both winds and currents respectively, and on morphological characteristics (Shteinman *et al*. 1997). This is especially true in shallow water bodies such as Coombabah Lake.

A previous study of seven cores within the lake using ^{210}Pb analysis (Frank & Fielding 2004) dated the top 2.5 cm of sediments in the lake at more than 150 years old. These results suggest there has been nil recent sedimentation and that surface sediments have been either reworked or removed. In addition, an influx of coarser-grained sand was observed from the seaward side of the lake, similar in nature to a tidal delta.

Studies by Gutteridge, Haskins and Davey Pty Ltd (2000) and Gallagher (2001) have both concluded that the lake shows evidence of increasing eutrophication. Gallagher (2001) measured historically high TOC and TP water concentrations, but low TN levels. The high TP concentrations were attributed to the sediments acting as a P source and it was concluded that the system was N limited. Extra N was likely to trigger further eutrophication.

Geochemical studies of three cores in Coombabah Lake by Fielding and Frank (2000) and heavy metal measurements by Dixon and Draper (1996) showed low background levels for Pb, Hg, As, Cd and Zn. However, these were not located in the eastern side of the lake near likely sources of metals from the upper Coombabah Creek (draining sections of the Pacific Highway and industrial estates), the STP or the Suntown landfill. It was noted in data supplied by Paice

(1996) that acidic conditions likely to transport soluble metals were recorded for water samples taken from a local drain near the sewage treatment plant.

The coastal wetlands surrounding Coombabah Lake have been mapped as having a high risk of containing acid sulfate soils and rich in iron sulfides of estuarine origin (Malcolm *et al*. 2002). Iron sulfide distribution and morphology in estuarine sediments reflect the processes and sedimentary environment during accumulation (Howarth 1979; Rabenhorst & Haering 1989; Wang & Morse 1996) and can affect oxidation behaviour (Bronswijk *et al*. 1993; Evangelou 1995; Bush & Sullivan 1996, 1998). The characteristics and reactivity of pyrite are known to vary significantly (van Dam & Pons 1973; Bertolin *et al*. 1995). Iron monosulfides (e.g. mackinawite, greigite) are far more reactive than framboidal pyrite and appreciable quantities of these highly reactive sulfides have been observed in some Australian estuarine sediments (Bush & Sullivan 1997, 1998). The status of iron sulfides within Coombabah Lake prior to this study was not known, nor was their potential role in contributing to any deoxgenation of the lake or making metals in the sediments potentially bioavailable.

Approach and methods

The study involved the collection and integration of collected cores from 53 sample grids incorporating the entire lake floor to characterise surface sediments of Coombabah Lake. Other aspects were the investigation of metal and sulfide risk and finally an assessment of sedimentation rates using shell radiocarbon dating.

Sediment sampling survey

To assess and characterise selected physico-chemical and biological parameters (Table 2-1) of the bottom sediments a systematic point sampling approach was undertaken (Caeiro *et al*. 2003). During seven survey periods (Table 2-2) in October 2004, a total of 265 surface sediment cores were collected in 53 sample grids (Figure 2-1) within Coombabah Lake. The scale and number of sample grids were chosen to represent the major regions of the lake sediments and ranged in size from 250x250 m (51), 500x250 m (1) and 750x250 m (1).

Table 2-1. Physico-chemical surface sediment parameters measured within Coombabah Lake

Physico-chemical parameter	Sample depth (cm) 0–1	0–2	2–5
Wet bulk density		X	X
Moisture content (wet and dry)		X	X
Porosity		X	X
Colour		X	X
Organic matter (LOI_{550})		X	X
Carbonate content (LOI_{950})		X	X
Chlorophyll *a* content	X		
Phaeopigment content	X		
Mineralogy		X	X
Grain size		X	X
$d_{(0.1)}$, $d_{(0.5)}$, $d_{(0.9)}$ (µm)			
% clay, silt and sand fractions (Wentworth size classes)			
Grain texture		X	X
% Organic C and % organic N		X	
% Organic N		X	
C:N		X	
$\delta^{13}C$		X	
$\delta^{15}N$		X	
Total phosphorous		X	X
1M $MgCl_2$ Extractable phosphorous		X	X
1M KCl Extractable ammonium		X	X

Sample grids were located using a hand-held GPS receiver (Magellan; Meridian Series). Sample surveys commenced at the marine entrance of the lake system 2 h before high tide period. This approach has the advantage of allowing the maximum sampling time from the sample vessel in addition to working at times of low tidal velocities. Samples were collected from a sample vessel using a cylindrical PVC coring tube (5 cm i.d., 40 cm length). Within each of the sample grids, five sediment cores were collected and divided into 'upper' (0–2 cm) and 'lower' (2–5 cm) depths before being homogenised and stored in dark conditions at <4^{o}C in the field. Typically samples were transported to the laboratory within 4 h after collection. Samples were stored frozen (-20^{o}C) until analysed.

Table 2-2. Coombabah Lake sample grid details and physical features

Grid	Latitude (centre) (°S)	Longitude (centre) (°E)	Date sampled	Vegetation dominance	Relative water depth	Low tide exposure	Adjoining land[a]	Adjoining creek[b]	Developed region	Approximate area per core (m^2)[c]
1	–27.91728600	153.3488241	19/10/04	Mangrove	Average	√	√	√	–	8750
3	–27.91503463	153.3477009	25/10/04	Mangrove–saltmarsh	Deep	–	√	√	–	1875
4	–27.91616032	153.3488331	19/10/04	Mangrove	Deep	–	√	√	–	8750
6	–27.91390894	153.3443312	16/10/04	Mangrove–saltmarsh	Average	√	√	–	√	625
7	–27.91390894	153.3465776	19/10/04	Mangrove–saltmarsh	Average	√	√	√	–	750
8	–27.91390894	153.3488310	19/10/04	Mangrove	Average	–	√	√	–	10000
9	–27.91390894	153.3510705	19/10/04	Mangrove	Average	–	√	-	–	9375
10	–27.91390894	153.3533169	19/10/04	Mangrove	Average	–	√	√	√	11875
11	–27.91390894	153.3555633	25/10/04	Mangrove	Average	√	√	–	√	750
13	–27.91165757	153.3443312	16/10/04	Mangrove–saltmarsh	Shallow	√	√	–	√	6250
14	–27.91165757	153.3465776	20/10/04	–	Average	√	–	–	–	12500
15	–27.91165757	153.3488310	20/10/04	–	Average	–	–	–	–	12500
16	–27.91165757	153.3510705	20/10/04	–	Average	–	–	–	–	12500
17	–27.91165757	153.3533169	20/10/04	–	Average	–	–	–	–	12500
18	–27.91165757	153.3555633	20/10/04	–	Average	√	–	–	–	12500
19	–27.91165757	153.3578098	26/10/04	Mangrove	Shallow	√	√	–	–	6250
20	–27.90940619	153.3443312	16/10/04	Swamp oak	Shallow	√	√	–	√	11250
21	–27.90940619	153.3465776	25/10/04	–	Average	√	–	–	–	12500
22	–27.90940619	153.3488310	26/10/04	–	Deep	√	–	–	–	12500
23	–27.90940619	153.3510705	25/10/04	–	Deep	–	–	–	–	12500
24	–27.90940619	153.3533169	25/10/04	–	Deep	–	–	–	–	12500
25	–27.90940619	153.3555633	26/10/04	–	Average	√	–	–	√	12500
27	–27.90715482	153.3443312	16/10/04	Swamp oak	Shallow	√	√	–	–	2500
28	–27.90715482	153.3465776	19/10/04	–	Average	√	–	–	–	12500
29	–27.90715482	153.3488310	19/10/04	–	Deep	√	–	–	–	12500
30	–27.90715482	153.3510705	19/10/04	–	Deep	–	–	–	–	12500
31	–27.90715482	153.3533169	19/10/04	–	Deep	–	–	–	–	12500
32	–27.90715482	153.3555633	19/10/04	–	Deep	–	–	–	–	12500

Table 2-2. Coombabah Lake sample grid details and physical features (continued)

Grid	Latitude (centre) (°S)	Longitude (centre) (°E)	Date sampled	Vegetation dominance	Relative water depth	Low tide exposure	Adjoining land[a]	Adjoining creek[b]	Developed region	Approximate area per core (m^2)[c]
33	–27.90715482	153.3578098	19/10/04	Mangrove	Average	√	√	√	–	6250
34	–27.90715482	153.3589330	25/10/04	Mangrove	Average	√	√	√	–	625
35	–27.90490344	153.3443312	15/10/04	Mangrove–saltmarsh	Shallow	√	√	–	√	625
36	–27.90490344	153.3465776	15/10/04	Mangrove–saltmarsh	Average	√	√	–	√	11250
37	–27.90490344	153.3488310	16/10/04	–	Average	√	–	–	–	12500
38	–27.90490344	153.3510705	19/10/04	–	Average	–	–	–	–	12500
39	–27.90490344	153.3533169	16/10/04	Mangrove	Average	–	–	–	–	7500
40	–27.90490344	153.3555633	25/10/04	Mangrove	Shallow	√	√	–	–	6250
41	–27.90490344	153.3578098	25/10/04	Mangrove–saltmarsh	Deep	–	√	–	–	8750
42	–27.90265207	153.3465776	16/10/04	Mangrove	Average	√	√	–	–	3125
43	–27.90265207	153.3488310	15/10/04	Mangrove	Average	√	√	–	–	11875
44	–27.90265207	153.3510705	16/10/04	Mangrove	Average	√	√	–	–	11875
45	–27.90265207	153.3533169	16/10/04	Mangrove	Deep	√	√	√	–	11250
46	–27.90265207	153.3555633	20/10/04	Mangrove	Deep	√	√	√	–	8125
47	–27.90265207	153.3578098	20/10/04	Mangrove–saltmarsh	Deep	√	√	√	–	4375
48	–27.90040069	153.3465776	15/10/04	Mangrove	Average	–	√	–	–	1875
49	–27.90040069	153.3488310	15/10/04	Mangrove	Deep	√	√	–	–	8125
50	–27.90040069	153.3533169	20/10/04	Mangrove	Shallow	–	√	√	–	6250
51	–27.90040069	153.3555633	20/10/04	Mangrove	Deep	–	√	√	–	12500
52	–27.90040069	153.3454544	15/10/04	Mangrove	Average	–	√	–	–	12500
54	–27.90040069	153.3533169	20/10/04	Mangrove	Average	√	√	√	–	3125
55	–27.89927500	153.3555633	25/10/04	Mangrove	Deep	–	√	√	–	12500
56	–27.89814932	153.3454544	15/10/04	Mangrove	Average	–	√	–	–	12500
58	–27.89814932	153.3555633	25/10/04	Mangrove	Deep	–	√	√	–	6250
59	–27.89814932	153.3578098	25/10/04	Mangrove	Deep	–	√	√	–	6250

where: [a] represents adjoining land, sample grids whose boundaries are connected to lake boundary/shorelines, [b] represents adjoining creeks, sample grids whose boundaries are connected to creek systems, [c] represents sediment area (excludes terrestrial regions within grid outline), – represents no and √ represents yes.

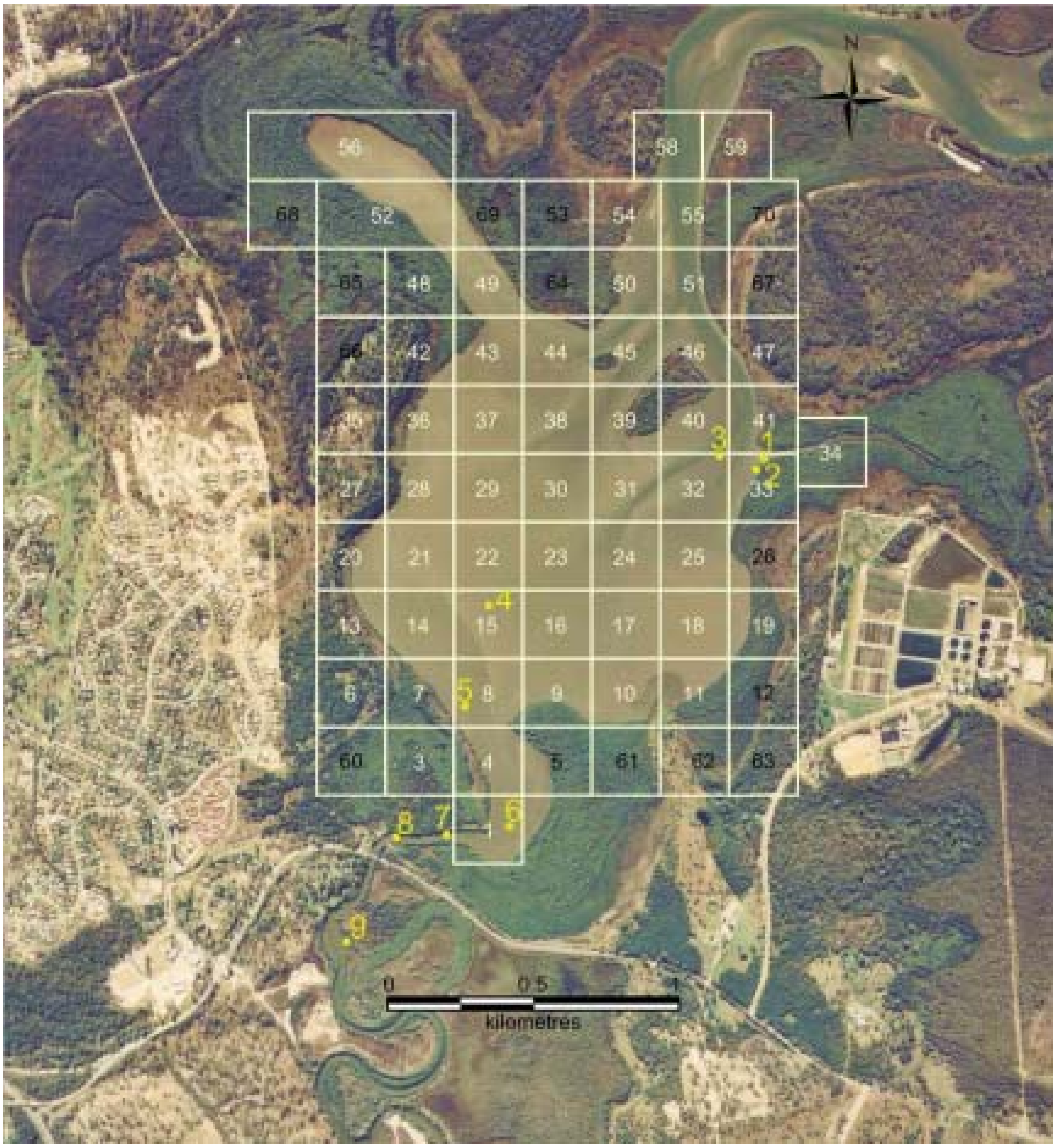

Figure 2-1. Coombabah Lake numbered sample grids

Note only sample grids within the boundaries of the lake were sampled (i.e. grids indicated with black numbers were not sampled). Yellow site numbers refer to case study sites investigating sulfides and metal risk.

General procedures and instrumentation

With the exception of chlorophyll *a* and phaeopigment concentrations and elemental carbon and nitrogen analyses, physico-chemical parameters were measured at both the upper and lower sediment depths (Table 2-3).

Wet bulk density

The wet bulk density of the sampled sediments was determined through the gravimetric syringe technique described by Boyce (1976), a variation of the core method described by Percival and Lindsay (1997). This method is used for soft sediments only and allows for the determination of porosity, wet bulk density, and water content on the same sample. Sediments were dried at 105°C for the determination of moisture content. The porosity of sediments was determined indirectly from wet bulk density and wet-water content values (Percival & Lindsay 1997).

The wet bulk density was calculated from the following equation:

$$\rho_{wb} = M_{ws} / V_{ws} \qquad \textbf{(Eq. 1.1)}$$

where: ρ_{wb} represents wet bulk density (g cm^{-3}),

M_{ws} represents the mass of wet sediment (g) and

V_{ws} represents the volume of wet sediment (cm^{3})

(Percival & Lindsay 1997).

Table 2-3. Sediment parameters, sample depths, abbreviations, symbols and analytical procedures

Sediment parameter	Sample /analysis depth (cm)			Abbreviation / symbol	Method	Reference
	0–1	0–2	2–5			
Wet bulk density	–	√	√	ρ_{wb}	Gravimetric syringe technique	Boyce 1976
Wet moisture content	–	√	√	Θ_W	Gravimetric syringe technique	Boyce 1976
Dry water content	–	√	√	Θ_D	Gravimetric syringe technique	Boyce 1976
Porosity percentage	–	√	√	ϕ	Gravimetric syringe technique	Percival & Lindsay 1997
Grain size	–	√	√	–	Mie scattering particle sizer	–
Texture	–	√	√	–	Sand, silt and clay triangle	Lewis & McConchie 1994
Mineralogy	–	√	√	–	X-ray diffraction (XRD)	O'Day *et al*. 2000
Colour	–	√	√	–	Munsell© colour chart	Lee & Cundy 2001
Organic matter content	–	√	√	LOI_{550}	Loss-on-ignition technique (LOI)	Heiri *et al.* 2001
Carbonate content	–	√	√	LOI_{950}	Loss-on-ignition technique (LOI)	Heiri *et al*. 2001
Chlorophyll *a* concentration	√	–	–	Chl *a*	Spectrophotometric analysis	Lorenzen 1967
Phaeopigment concentration	√	–	–	Phaeo	Spectrophotometric analysis	Lorenzen 1967
Total phosphorous concentration	–	√	√	TP	Molybdate blue method	APHA 1998
1M $MgCl_2$ extractable phosphorous concentration	–	√	√	E–P	Molybdate blue method	APHA 1998
1M KCl extractable ammonium concentration	–	√	√	E–NH_4^+	Phenate method	APHA 1998
Percentage of elemental carbon	–	√	–	% C	Mass spectroscopy–elemental analyser	–
Carbon isotopic fractionation	–	√	–	$\delta^{13}C$	Mass spectroscopy–elemental analyser	–
Percentage of elemental nitrogen	–	√	–	% N	Mass spectroscopy–elemental analyser	–
Nitrogen isotopic fractionation	–	√	–	$\delta^{15}N$	Mass spectroscopy–elemental analyser	–

Moisture (water) content

Dry and wet moisture (water) contents were determined by a direct gravimetric method, where individual wet sediment samples were placed in a pre-weighed sample boat. The weight of the wet sediment sample and weigh boat were recorded. The weigh boats were then placed in an oven and dried at 105^{o}C to a constant weight (typically 12 h). Weigh boats containing sediments were then allowed to cool to room temperature for 2 h in a desiccator then re-weighed. Moisture content was calculated using the following equation:

$$\Theta_w = M_w - M_{ws} \times 100 \qquad \textbf{(Eq. 2.2)}$$

$$= M_{ws} - M_{ds} / M_{ws} \times 100 \qquad \textbf{(Eq. 2.3)}$$

similarly,

$$\Theta_d = M_{ws} - M_{ds} / M_{ds} \times 100 \qquad \textbf{(Eq. 2.4)}$$

where: Θ_w represents wet-water content (%),

Θ_d represents dry-water content (%),

M_{ws} represents mass of wet sediment (g) and

M_{ds} represents mass of dry sediment (g) (Boyce 1976).

Porosity

Sediment porosity was determined indirectly from wet bulk density and wet–water content results:

$$\phi = 100 / \rho_w \times (M_{ws} - M_{ds}) / V_{ws} \qquad \textbf{(Eq. 2.5)}$$

$$= \rho_{wb} \times \Theta_w \qquad \textbf{(Eq. 2.6)}$$

where: ϕ represents porosity (%), ρ_w represents density of water (1.0g cm^{-3}), M_{ws} represents mass wet sediment (g), M_{ds} represents mass dry sediment (g), V_{ws} represents volume of wet sediment (cm^3), ρ_{wb} represents wet bulk density (g cm^{-3}) and Θ_w represents wet-water content (%) (Percival & Lindsay 1997).

Sediment colour

The colour of oven-dried sediments (105^{o}C, 12 h) was quantified using international system, Munsell® soil colour charts.

Organic carbon and carbonate content

Determination of the weight percent organic carbon and carbonate content in the sampled sediments was achieved by the loss-on-ignition (LOI) technique based on the sequential heating of samples. Oven-dried sediment samples (105°C, 12 h) (~2g) dried to a constant weight were transferred to a desiccator, allowed to cool (2 h) to room temperature and weighed. Samples were then transferred to a muffle furnace (SA Pty Ltd SEM muffle furnace/Brainchild Electronic Co. BTC-9090, 550^{o}C, 4 h) before being allowed to cool in a desiccator to room temperature and being re-weighed.

The LOI_{550} was calculated using the following equation:

$$LOI_{550} = ((DW_{105} - DW_{550}) / DW_{105}) \times 100 \quad \textbf{(Eq. 2.7)}$$

where: LOI_{550} represents LOI at 550°C (%), DW_{105} represents the dry weight of the sample before combustion (g) and DW_{550} represents the dry weight of the sample after heating to 550°C (g) (Heiri *et al.* 2001). Following the determination of LOI_{550}, the samples were re-entered into the muffle furnace (950°C, 4 h) to determine the carbonate content (LOI_{950}) The crucibles were then transferred to a desiccator and allowed to cool to room temperature before the determination of their final weights.

LOI_{950} was calculated as:

$$LOI_{950} = ((DW_{550} - DW_{950}) / DW_{105}) \times 100 \quad \textbf{(Eq. 2.8)}$$

where: LOI_{950} represents the LOI at 950°C (%), DW_{550} represents the dry weight of the sample after combustion of organic matter at 550°C (g), DW_{950} represents the dry weight of the sample after heating to 950°C (g), and DW_{150} represents again the initial dry weight of the sample before the organic carbon combustion (g) (Heiri *et al.* 2001).

Chlorophyll *a* and phaeopigment

Wet sediment samples designated for the determination of chl *a* and phaeo concentrations were homogenised and frozen (-20°C) in foil-wrapped glass scintillation vials. Frozen samples were then freeze-dried (CHRIST Alpha 1-4; 0.200 mbar, 12 h). Chl *a* and phaeo were then extracted in 4 ml of 90% acetone, agitated, before samples were stored in darkness (4°C, 24 h) with repeated agitation (to aid in the extraction of the pigments). Immediately prior to the determination of chl *a* and phaeo concentrations, samples were centrifuged (Hettich Zentrifugen Universal 16A, 10 min, 2500 rpm), to ensure absorption at 750 nm less than 0.005 (750 nm is a measure of the clarity of the sample; samples were re-spun if the extinction was greater than 0.005).

The supernatant was transferred into a glass cuvette and absorbance was measured at 665 and 750 nm. Samples were then acidified (2 drops 10% HCl) and the absorbance remeasured at 665 and 750 nm. Chl *a* and phaeo concentrations in the supernatant are expressed in µg chl *a* g^{-1} and µg phaeo g^{-1} respectively.

Chl *a* and phaeo concentrations were determined by spectrophotometry (Pharmacia Biotech visible spectrophotometer Novaspec II) following the equations:

$$\text{Chl } a\ (\mu\text{g g}^{-1}) = A \times K \times \left(\left(665_O - 750_O \right) - \left(665_A - 750_A \right) \right) \times v / \left(m_s \times l \right) \qquad \textbf{(Eq. 2.9)}$$

$$\text{Phaeo}\ (\mu\text{g g}^{-1}) = A \times K \times \left(\left(R\left[665_O - 750_O \right] \right) - \left(665_A - 750_A \right) \right) \times v / \left(m_s \times l \right) \qquad \textbf{(Eq. 2.10)}$$

(modified from Lorenzen 1967).

where: *A* represents the absorption coefficient for chlorophyll (11.0), *K* represents a factor to equate the reduction in absorbency to initial chlorophyll concentrations (2.43), 665_O represents the extinction at 665 nm before acidification, 665_A represents the extinction at 665 nm after acidification, 750_O represents the extinction at 750 nm before acidification, 750_A represents the extinction at 750 nm after acidification, v represents volume of acetone extract (ml), m_s represents mass of sediment sampled, and l represents the path length of cuvette (cm).

Sediment grain size and texture

Sediment grain size analysis was achieved through laser diffraction spectroscopy (Mastersizer 2000, Malvern Instruments Ltd, Mie scattering; size range: 0.020–2000.00 μm, obscuration between 10–20% accompanied with integrated software Mastersizer 2000, Malvern Instruments Ltd, Version 5.22). Grain sizes were determined from volumetric particle distributions measured on wet sediment samples. During sample storage, drying of the sediment before particle size analysis was avoided as it is very difficult, if not impossible, to redisperse the finest silt- and clay-sized particles after sediments have been allowed to dry.

The injection of samples was performed through the accessory Hydro 2000MU sample dispersion unit using deionised water (pump speed 2000, Malvern Instruments Ltd). Samples were ultrasonically pre-treated (ultrasonic displacement: 20.00, 1.5 min) before analysis through the accessory Hydro 2000MU sample dispersion unit (Malvern Instruments Ltd). Sonication was employed to enhance the dispersion of aggregates, as some sediments may contain organic matter, salts, iron oxides or carbonate coatings that bind particles together. Sediment texture for the sampled lake sediments was determined through the use of a gravel-free sand, silt and clay triangular diagram (Lewis & McConchie 1994).

Sediment mineralogy

Sediment samples designated for mineralogy determination were air-dried to facilitate fragile clay species not being destroyed. The dried samples (5 cm^3) were then ground to a grain size of <5 μm through a shatter box ring mill equipped with a zirconia grinding head (Rocklabs Ltd) before X-ray diffraction (XRD) analysis.

Whole sediment mineralogy samples were analysed using a Bruker D8 Advance X-ray diffractometer equipped with a graphite monochromator, copper target and scintillation counter (detector) under the following conditions: 2 to 90 degrees 2–theta, 0.02 degree step size or increment, 1 second per step (time per step), 1 h 13 min scan time variable slits: 20 mm anti-scatter slit, 20 mm divergence slit, 40 kV, 30 mA.

Traces were processed using the Diffrac$^{(plus)}$ evaluation package (2004) and PDF2 (Powder Diffraction File) (2004). The raw data files were analysed through the use of EVA software (SOCABIM©).

Sediment percent organic C and N content, C:N ratio and $\delta^{13}C$ and $\delta^{15}N$ of organic matter

Prior to C and N analysis, 2 g of sediment from each sample grid were ground (<5 µm) and treated with 5 ml 1M HCl to remove carbonates. Samples for C and N ratios and isotope analysis were dried at 40°C (temperature low enough that would not dramatically alter the nutrient status of the sediment) to a constant weight before analysis. Samples were analysed through mass spectroscopy (Isoprime, GV Instruments) equipped with elemental analyser (Eurovector 3000), with chromium oxide, silvered cobaltous/cobaltic oxide, silver metal combustion tube and working temperature of 1050°C and reduced copper metal (wire form) reduction tube with working temperature of 610°C and magnesium perchlorate drying tube.

Analyses of % C and % N were performed on the upper surface sediments only. ANU sucrose and ambient air, IAEA-305a were used as primary standards for N and C respectively. The elemental standard was acetanilide. The relative abundances of the heavy and light isotopes of C and N are expressed in standard delta notation (per mil, ‰), with negative values indicating depletion of the heavier isotope relative to a standard reference sample. $\delta^{13}C$ and $\delta^{15}N$ values were calculated according to the following formula:

$$\delta X \text{ (in ‰)} = ((R_{sample}/R_{standard}) / R_{standard}) \times 1000 \qquad \textbf{(Eq. 2.11)}$$

where X represents either ^{13}C or ^{15}N and R represents either $^{13}C/^{12}C$ or $^{15}N/^{14}N$ based on C and N standards respectively.

Total phosphorous

Total phosphorous (TP) concentrations on sediment samples were determined through modified methods of Pardo *et al*. 2004 and the APHA Handbook method 4500–P.D. Stannous chloride (APHA 1998). Oven-dried sediment samples (105°C, 12 h) (Contherm digital series oven) were homogenised and ground before ~0.200 g samples were transferred to a muffle furnace (SA Pty Ltd, SEM muffle furnace/Brainchild Electronic Co. BTC-9090, 450°C, 3 h) for the calcination of individual samples. Samples were allowed to cool in a desiccator to room temperature and transferred into glass centrifuge tubes before a simple extraction (16 h) with 3.5 mol l^{-1} HCl was performed within a water bath set at laboratory temperature (Thermoline Scientific Equipment Pty Ltd with Brainchild BTC-9090). Samples were then centrifuged (Hettich Zentrifugen Universal 16A, 3000 rpm, 10 min) before spectrophotometric determination was carried out using the modified stannous chloride method on diluted samples (1:10) at 690 nm.

Diluted samples underwent the addition of 2 drops of 3M NaOH until a pink colour developed (typically ~1.25 ml), H_2SO_4 to reverse the pink colouration (typically ~10–15 µl), 440 µl ($(NH_4)_6Mo_7O_{24.4}H_2O$) reagent and 1 drop $SnCl_2$. After the addition of $SnCl_2$ the absorbance was measured by spectrophotometry (Pharmacia Biotech visible spectrophotometer Novaspec II) after colour development (10–12 min).

The precision of the analysis was monitored by running triplicates every ~5 samples and was generally <5% relative standard deviation (RSD). Standard deviation values were utilised for the determination of the method detection limit. Total P concentrations were reported in µg g^{-1} dry weight (dry wt) of the sediment.

1M $MgCl_2$ Extractable phosphorous

1M $MgCl_2$ extractable phosphorous concentrations on sediment samples were determined through the modified methods of APHA Handbook method 4500-P D. Stannous chloride (APHA 1998).

Frozen sediment samples (-20°C) were thawed and centrifuged (Hettich Zentrifugen Universal 16A, 3000 rpm, 3 min) to reduce the presence of suspended sediments within the supernatant. Five ml of supernatant was then transferred from the centrifuge tube to clean sample acid-washed (10% HCl) vials before spectrophotometric determination at 690 nm.

Samples underwent the addition of reagents in the same sequence and fashion as that of the TP analysis.

The precision of the analysis was monitored by running triplicates every ~5 samples and was generally <5% RSD. Standard deviation values were utilised for the determination of the method detection limit. E-P concentrations were reported in µg g^{-1} dry wt of the sediment.

1M KCl Extractable ammonium

Extractable ammonium (E-NH_4^+) concentrations on sediment samples were determined through modified methods of the APHA Handbook method 4500–NH_3 F. Phenate method (APHA 1998).

Frozen sediment samples (-20°C) were thawed and centrifuged (Hettich Zentrifugen Universal 16A, 3000 rpm, 3 min) to reduce the presence of suspended sediments within the supernatant. Five ml of supernatant was then transferred from the centrifuge tube to clean sample acid-washed (10% HCl) vials before spectrophotometric determination at 640 nm using the phenate method on diluted samples (typically 1:2, 1:4 and 1:6).

Diluted samples underwent the addition of 200 µl C_6H_6O, 200 µl $Na_2[Fe(CN)_5NO]\cdot 2H2O$ and 500 µl oxidising solution. Samples were covered with paraffin film and allowed to stand at room temperature (22–27°C) for 1 h in subdued light to allow colour development (colour stable for 24 h). After colour development, absorbance was measured by spectrophotometry (Pharmacia Biotech visible spectrophotometer Novaspec II) at a wavelength of 640 nm.

The precision of the analysis was monitored by running triplicates every ~5 samples and was generally <5% RSD. Standard deviation values were utilised for the determination of the method detection limit. Extractable NH_4^+ concentrations were reported in µg g^{-1} dry wt of the sediment.

Metal and sulfides

Scoping study

To assess if sulfides were likely to be a sediment issue potentially affecting water quality in Coombabah Lake, the levels of sulfur and iron sulfide present in lake sediments were determined in a scoping study. Sediment samples taken were a composite of five individual cores taken from the surface (0–2 cm deep) of 10 locations on the lake (Figure 2-2).

The same samples collected for the assessment of sulfides—a composite of five individual cores taken from the surface (0–2cm deep)—were also tested for metals. Using the Department of Natural Resources, Mines and Water method EWS–S14 (Jeffrey 2002), metals tested included Al, As, Ca, Co, Cr, Co, Fe, K, Mg, Mn, Mo, Na, Ni, Pb and Zn. Such strong acid digestion methods dissolve almost all elements that could become “environmentally available”. Such a procedure, however, cannot digest crystalline silicates without the use of HF. Nevertheless, this and other similar methods are commonly used to determine the near metal content of sediment samples (USEPA 1986; Burton *et al*. 2005a).

As an initial assessment of risk of metal bioaccumulation, this data was compared with Australian sediment quality guidelines (ANZECC/ARMCANZ, 2000). Iron sulfide S was compared with the action levels specified in Queensland Government technical guidelines for the management of acid sulfate soils (Dear *et al.* 2002). Iron sulfide S values were also used as a guide to indicate where monosulfide risk is potentially the greatest in Coombabah Lake.

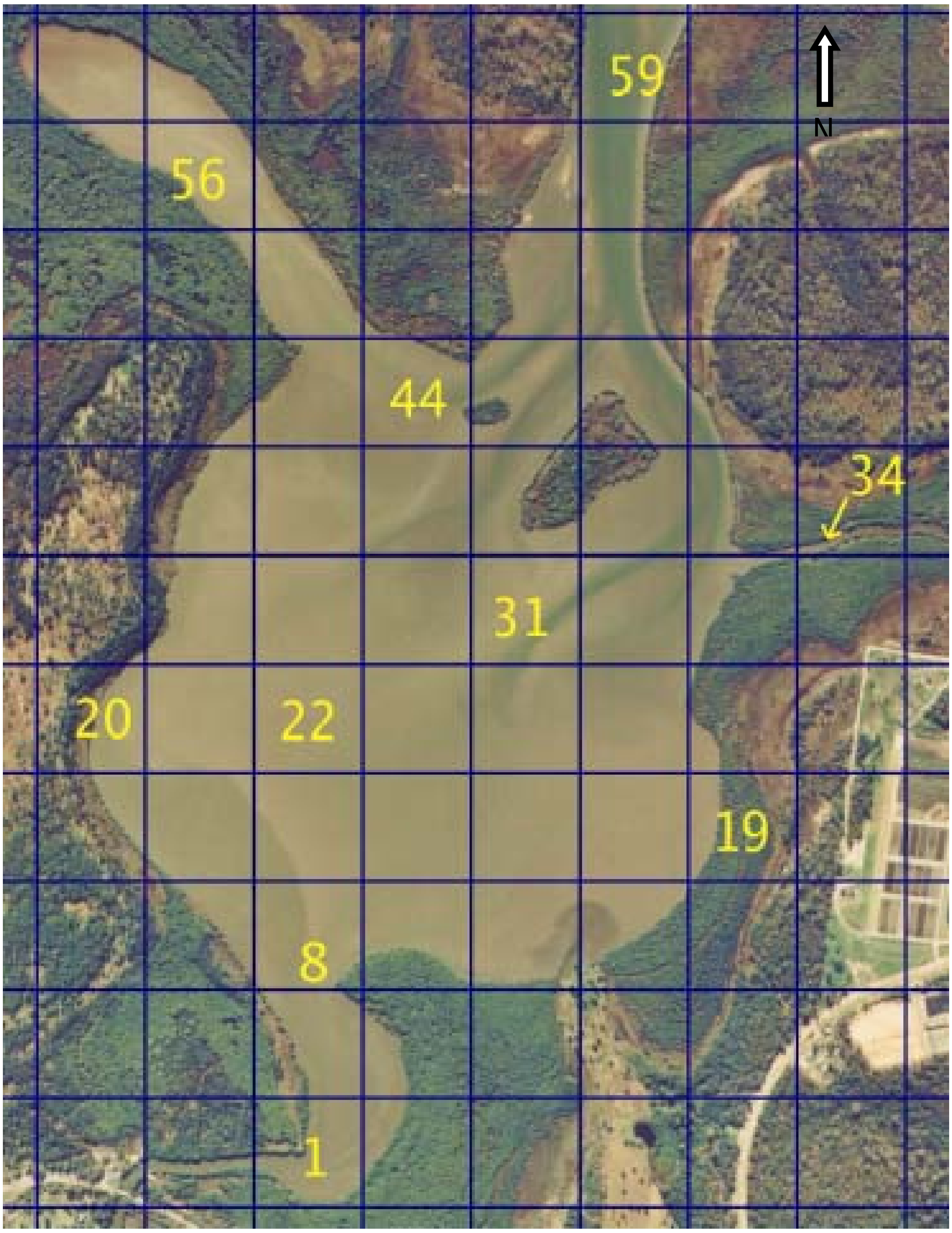

Figure 2-2. Location of surface sediment samples (250 m x 250 m grid), Coombabah Lake

Total sulfur was determined by Department of Natural Resources, Mines and Water method EWS-S14 (Jeffrey 2002), involving a digest of microwave triacids, nitric, perchloric and peroxide and ICP determination. Iron sulfide S (mainly pyrite and sulfur) was determined by the chromium reducible sulfur method (Sullivan *et al.* 2000).

Monosulfide geochemistry STP Creek case study

Monosulfides is the term given to the highly reactive iron sulfide minerals found in soils and sediments that have the approximate formula ‘FeS’ and which are soluble in hydrochloric acid (as opposed to iron disulfides such as pyrite that aren’t appreciably soluble in hydrochloric acid). They are also commonly described as acid volatile sulfides (AVS).

To assess the risk of resuspended monosulfides impacting on dissolved oxygen levels in lake waters, three sediment cores were sectioned into 2 cm or 4 cm depths at or near the mouth of STP Creek (Sites 1 to 3 in Figure 2-1). Each section was analysed for pH, Eh (expressed versus SHE), water content and AVS. AVS was determined by the cold diffusion technique (Hsieh *et al*. 2002), using ascorbic acid to prevent Fe(III) interferences.

To assess if the monosulfides present in the sediments could lower the level of dissolved oxygen of lake water, immediate sediment oxygen demand (ISOD) was also determined on the sediments down to 10 cm depth for each site. ISOD is a useful unpublished technique used extensively in estuarine environments in New South Wales (L. Sullivan, pers.com.). A quantity of wet sediment (10 g oven dry weight equivalent) was placed into 200 ml of fresh sea water (with an oxygen content of 7.92 mg l^{-1}) and then swirled to disperse the sediment into the water column. After 5 minutes (by which time the dissolved oxygen levels are quite stable) the dissolved oxygen content was measured and the ISOD value determined in grams of oxygen consumed per gram of dry sediment for the 5-minute period.

Integrated sulfur, iron and trace metal case study

Intact sediment cores (0–30 cm depth and internal diameter 10 cm) were collected from selected sites in Coombabah Lake. Four cores (Sites 4, 5, 6 and 9) were characterised with the objective of describing the geochemical interactions of sulfur, iron and trace metals and examined trends from the lake upstream to the reaches of upper Coombabah Creek (Figure 2-1). This was also a means of undertaking a more comprehensive risk assessment of metal contamination and monosulfide risk in a zone identified by the scoping study as having some indication of higher risk within the system.

The cores were sectioned into eight depth segments. As a quality assurance measure, duplicate analyses were performed on 25% of samples. Therefore, a suite of analyses were performed on a total of 40 samples (4 cores x 8 depth segments = 32 samples; 32 samples + 25% QA duplicates = 40 samples), including pH, Eh, pore-water Fe(II), pore-water S(-II), pore-water SO_4, acid-volatile sulfide, elemental sulfur, pyrite, 1M HCl-extractable metals, total C, total S and total metals (by strong acid digestion). This substantial dataset provides a thorough characterisation of sediment geochemistry and included a state-of-the-art speciation of reduced inorganic sulfur.

Pore-water and solid-phase analyses of samples followed the specific extraction, preservation and analytical protocols described in (Burton *et al*. in press, a).

Australian sediment quality guidelines are based on total concentrations (ANZECC/ARMCANZ, 2000). The further assessment of metal bioaccumulation risk in estuarine sediments using 1M HCl-extractable metals (as recommended by ANZECC/ARMCANZ, (2000) and AVS data is based on the approach applied by Burton *et al*. (2005b) to Moreton Bay sediments. Briefly, this approach involved two further assessments:

1. Sediments with a molar excess of AVS over the molar sum of 1M HCl-extractable Cu, Cd, Ni, Pb and Zn are expected to exhibit very low pore-water metal activities, and thus low bioavailability to benthic organisms. In contrast, for sediments where AVS is less than the molar sum of SEM, pore-water metal concentrations may become high and cause toxicity to benthic biota if other non-sulfidic binding phases are not present. This assessment relies on the theory that many trace metals precipitate in a 1:1 molar ratio as highly insoluble metal sulfide minerals when exposed to reactive sulfide, estimated in sediments as AVS.
2. %AVS/Fe was calculated to accommodate seasonal metal sulfide interactions in surficial sediments. When AVS constitutes less than 20% of reactive 1M HCl-extractable Fe (i.e. %AVS/Fe <20), the sediment/ pore-water system is dynamic with respect to trace metal mobility. Under these conditions, AVS formation/loss result in significant seasonal shifts in trace metal fluxes across the sediment–water interface and in sediment trace metal concentrations. Conversely, when greater than 20% of 1M HCl-extractable Fe on a molar basis was bound within AVS minerals (i.e. %AVS/Fe >20), there is sufficient sulfide to effectively trap trace metals within the sediments (Cooper & Morse 1998). Under these conditions, temporal variations in AVS do not necessarily result in changes to metal bioavailability as the system is well ‘buffered’ against oxidation of trace metal sulfide minerals. This dynamic behaviour is in need of further research.

Dating of shells

In an attempt to assess historic lake sedimentation rates, shells samples were taken from lake sediments and adjacent mangrove ponds for conventional radio carbon dating. The shell species were identified and samples tested for recrystallisation. Shells were cleaned in an ultrasonic bath, shell exteriors were acid washed using dilute HCl and rinsed and dried. The samples were crushed prior to hydrolysis and dating.

The Waikato Laboratory determined C^{14} activity through the measurement of beta particles. Samples were converted to benzene through hydrolysis of lithium carbide and catalytic trimerisation of acetylene. Residual radiocarbon activity was measured using Perkin Elmer 1220 "Quantulus" Liquid Scintillation (LS) spectrometers. Corrections were made for delta C^{13}.

Results and discussion

The sediments of estuarine environments are important recipients of all materials discharged into them, including from both natural and anthropogenic sources. The collection and storage of sediments may also leave these systems potentially vulnerable to pollution (Jickells & Rae 1997). The critical issue with storage of contaminants in intertidal systems is their potential for re-release which can occur through biological, chemical and physical processes (Davis Jr. 1997).

Sediments are complex deposits of inorganic particles, organic matter and adsorbed and dissolved constituents and can be characterised in terms of several readily measured physical properties that reflect their provenance and depositional environment. Depositional and postdepositional processes, as well as mineral composition, ultimately influence the texture, bulk properties and chemical characteristics of sediments (Penthick 1984). Results are presented and discussed below under three main themes: firstly, the results of the sediment sampling survey provides a physicochemical overview of lake sediments; secondly, the results of risk assessments of the metals and sulfides present in the sediments are discussed; and finally the results of radiocarbon dates for shell samples are presented and discussed.

Sediment sampling survey

The physico-chemical properties of surface sediments of Coombabah Lake were measured at depths of 0–2 cm and 2–5 cm, upper and lower depths respectively (Tables 2-4 and 2-5).

Table 2-4. General characteristics of sediment collected from *upper* surface sediments within Coombabah Lake

Sample grid	ρ_{wb} (g cm^{-3})	Θ_D (%)	Θ_W (%)	ϕ (%)	Sediment grain size distribution (µm)			Wentworth size class (%)			Texture	Colour	Chl a (µg g^{-1} dry wt)	Phaeo (µg g^{-1} dry wt)	LOI_{550} (%)	LOI_{950} (%)
					$d_{(0.1)}$	$d_{(0.5)}$	$d_{(0.9)}$	Clay	Silt	Sand						
1	1.53	95.10	48.75	74.75	2.54	11.21	139.54	7.75	79.38	12.87	Sandy silt	Greyish brown	1.18	6.93	9.01	1.79
3	1.44	118.28	54.19	78.24	2.58	12.33	138.25	7.61	76.89	15.50	Sandy silt	Light brownish grey	0.75	11.03	7.45	1.36
4	1.52	87.99	46.81	71.17	2.54	14.15	132.41	7.87	69.99	22.14	Sandy silt	Greyish brown	2.54	7.66	8.70	1.89
6	1.49	104.10	51.00	75.96	2.63	17.47	165.74	9.29	64.06	26.65	Sandy silt	Greyish brown	0.96	9.40	7.79	1.41
7	1.53	96.22	49.04	75.24	2.51	12.63	117.25	8.02	76.66	15.32	Sandy silt	Greyish brown	1.48	7.94	7.10	1.96
8	1.67	65.43	39.55	66.02	2.58	16.47	164.32	7.73	83.82	8.45	Silt	Greyish brown	1.38	7.86	4.87	1.95
9	1.62	75.19	42.92	69.40	2.92	22.24	156.61	6.43	69.02	24.55	Sandy silt	Grey	1.28	5.61	7.06	2.04
10	1.47	104.54	51.11	74.89	2.57	16.23	489.79	8.06	82.27	9.67	Silt	Greyish brown	2.23	11.29	4.59	4.00
11	1.40	125.78	55.71	78.14	2.84	12.72	111.67	6.46	82.31	11.23	Sandy silt	Greyish brown	2.88	13.57	6.39	0.96
13	1.45	133.16	57.11	82.61	2.36	10.49	98.23	8.89	80.47	10.64	Sandy silt	Grey	0.53	8.89	7.11	1.10
14	1.74	49.25	33.00	57.45	3.63	44.99	147.33	5.52	58.50	35.98	Sandy silt	Greyish brown	1.07	9.21	5.34	0.83
15	1.58	79.73	44.36	70.18	3.12	27.87	113.59	5.92	69.76	24.32	Sandy silt	Greyish brown	3.28	15.97	3.84	1.04
16	1.63	71.77	41.78	68.03	3.08	41.53	189.96	6.40	61.56	32.04	Sandy silt	Greyish brown	3.73	15.31	4.56	0.93
17	1.53	94.67	48.63	74.26	3.16	28.97	167.74	3.54	73.36	23.10	Sandy silt	Greyish brown	3.20	12.44	6.35	0.95
18	1.49	104.89	51.19	76.36	3.07	23.97	229.13	5.94	72.17	21.89	Sandy silt	Grey	2.87	13.52	6.05	0.97
19	1.43	125.94	55.74	79.48	2.83	16.41	299.91	6.65	79.44	13.91	Sandy silt	Greyish brown	0.43	10.29	7.51	0.93
20	1.65	66.36	39.89	65.94	2.80	16.36	187.07	6.68	75.39	17.93	Sandy silt	Greyish brown	2.56	11.16	3.65	1.15
21	1.55	78.02	43.83	68.15	3.28	41.48	268.29	6.01	68.16	25.83	Sandy silt	Greyish brown	4.87	15.11	3.13	1.20
22	1.64	77.20	43.57	71.27	2.99	36.10	189.23	6.74	65.88	27.38	Sandy silt	Greyish brown	4.80	17.48	4.18	1.20
23	1.67	67.01	40.13	66.85	3.61	91.94	311.63	5.31	64.18	30.55	Sandy silt	Greyish brown	3.20	14.47	3.92	0.87
24	1.60	78.09	43.85	70.02	2.74	15.82	117.49	7.03	73.83	19.14	Sandy silt	Greyish brown	2.45	13.32	4.68	1.07
25	1.58	92.42	48.03	76.03	3.40	49.97	295.48	5.64	67.66	26.70	Sandy silt	Greyish brown	1.60	11.61	4.26	0.99
27	1.49	97.88	49.46	73.70	2.11	10.74	452.08	10.56	80.41	9.03	Silt	Greyish brown	0.85	11.25	4.79	0.82
28	1.67	50.49	33.55	56.03	3.16	29.98	222.12	5.75	68.73	25.52	Sandy silt	Greyish brown	4.92	17.86	4.17	0.79
29	1.71	56.93	36.28	62.15	4.05	116.49	326.81	4.66	61.97	33.37	Sandy silt	Greyish brown	5.41	18.32	3.44	1.17
30	1.66	51.94	34.18	56.77	4.71	176.49	363.81	3.94	65.57	30.49	Sandy silt	Greyish brown	3.82	13.41	3.44	0.88
31	1.74	54.84	35.42	61.46	4.50	158.86	346.21	4.08	64.66	31.26	Sandy silt	Greyish brown	4.15	15.18	3.20	0.97
32	1.57	77.82	43.76	68.63	3.03	34.06	225.91	6.18	58.57	35.25	Sandy silt	Greyish brown	0.85	5.52	5.40	0.91
33	1.52	93.30	48.27	73.32	3.21	86.33	342.45	6.09	41.70	52.21	Silty sand	Greyish brown	2.65	12.44	7.20	0.97
34	1.34	120.22	54.59	73.16	3.43	162.82	388.02	5.49	53.69	40.82	Sandy silt	Greyish brown	0.53	10.44	7.91	1.15
35	1.93	30.83	23.57	45.38	4.05	225.33	458.39	4.81	26.52	68.67	Silty sand	Greyish brown	1.17	10.51	1.10	0.96

Table 2-4. General characteristics of sediment collected from *upper* surface sediments within Coombabah Lake (continued)

Sample grid	ρ_{wb} (g cm^{-3})	Θ_D (%)	Θ_W (%)	ϕ (%)	Sediment grain size distribution (µm)			Wentworth size class (%)			Texture	Colour	Chl a (µg g^{-1} dry wt)	Phaeo (µg g^{-1} dry wt)	LOI_{550} (%)	LOI_{950} (%)
					$d_{(0.1)}$	$d_{(0.5)}$	$d_{(0.9)}$	Clay	Silt	Sand						
36	1.73	52.82	34.57	59.87	5.54	259.77	511.48	3.24	35.60	61.16	Silty sand	Greyish brown	3.73	14.49	3.03	1.44
37	1.73	55.49	35.69	61.68	4.49	161.62	367.68	4.50	28.18	67.32	Silty sand	Greyish brown	2.98	15.49	3.31	1.27
38	1.82	46.22	31.61	57.42	6.89	186.58	351.11	2.73	18.73	78.54	Silty sand	Greyish brown	6.04	20.04	3.15	0.90
39	1.73	50.48	33.55	57.92	8.32	170.96	284.80	2.46	16.32	81.22	Silty sand	Greyish brown	4.48	15.75	2.79	1.22
40	1.90	44.85	30.96	58.75	11.13	214.85	375.45	1.98	14.09	83.93	Silty sand	Greyish brown	4.14	14.08	2.51	0.89
41	1.89	41.01	29.08	54.92	7.80	208.16	339.80	2.42	14.99	82.59	Silty sand	Greyish brown	5.44	18.80	1.69	1.23
42	1.93	31.93	24.20	46.81	5.16	199.49	456.72	3.50	26.19	70.31	Silty sand	Greyish brown	4.26	13.50	1.78	1.21
43	1.60	63.57	38.86	62.00	3.09	28.80	249.13	3.49	25.89	70.61	Silty sand	Greyish brown	1.06	11.85	5.55	1.03
44	1.66	61.87	38.22	63.46	3.45	75.52	205.24	5.71	37.36	56.93	Silty sand	Greyish brown	2.35	15.04	3.96	0.83
45	1.69	60.55	37.72	63.68	5.94	177.06	339.13	3.21	20.09	76.60	Silty sand	Grey	2.02	10.28	4.10	0.41
46	1.96	33.86	25.30	49.51	6.60	214.22	341.21	2.77	15.24	81.99	Silty sand	Greyish brown	2.99	9.43	1.78	5.56
47	1.73	60.95	37.87	65.42	2.30	21.15	358.39	9.37	54.28	36.35	Sandy silt	Greyish brown	1.06	8.78	3.29	0.49
48	1.71	58.62	36.96	63.37	4.60	95.94	271.68	3.91	38.99	57.10	Silty sand	Greyish brown	2.44	10.02	4.27	0.56
49	1.71	57.54	36.52	62.60	3.44	43.10	233.51	12.91	46.41	40.68	Sandy silt	Greyish brown	1.49	8.24	3.30	0.07
50	1.88	31.08	23.71	44.63	128.48	230.95	353.42	0.90	7.09	92.01	Sand	Greyish brown	2.88	9.14	1.43	0.16
51	1.83	24.52	19.69	36.00	171.56	261.13	389.91	0.06	1.52	98.42	Sand	Greyish brown	0.85	6.27	0.70	0.69
52	1.66	67.46	40.29	66.91	3.37	21.99	233.94	1.28	67.07	31.65	Sandy silt	Grey	2.66	10.05	5.41	1.58
54	1.99	30.08	23.12	45.93	23.95	211.33	334.15	1.57	9.95	88.48	Silty sand	Grey	2.12	7.32	1.94	0.57
55	1.72	52.66	34.50	59.35	3.31	147.45	335.66	5.79	29.86	64.35	Silty sand	Grey	0.74	9.87	2.32	7.97
56	1.54	93.81	48.40	74.35	3.00	16.44	167.32	6.00	71.03	22.97	Sandy silt	Grey	2.65	11.55	7.02	1.30
58	2.27	25.53	20.33	46.14	148.72	232.01	339.34	0.29	4.31	95.40	Sand	Grey	4.78	14.67	0.77	4.24
59	1.90	38.72	27.91	53.02	5.08	193.70	341.94	3.32	21.82	74.86	Silty sand	Dark greyish brown	14.40	34.49	0.64	0.79

Table 2-5. General characteristics of sediment collected from *lower* surface sediments within Coombabah Lake

Sample grid	ρ_{wb} (g cm^{-3})	Θ_D (%)	Θ_W (%)	ϕ (%)	Sediment grain size distribution (µm)			Wentworth size class (%)			Texture	Colour	LOI_{550} (%)	LOI_{950} (%)
					$d_{(0.1)}$	$d_{(0.5)}$	$d_{(0.9)}$	Clay	Silt	Sand				
1	1.50	103.11	50.77	75.97	2.58	11.15	140.79	7.22	87.09	5.69	Silt	Grey	7.33	1.93
3	1.50	94.88	48.69	73.25	2.02	10.26	114.91	11.14	71.26	17.60	Sandy silt	Grey	6.51	1.64
4	1.51	94.52	48.59	73.34	2.24	10.24	111.89	9.63	71.74	18.63	Sandy silt	Grey	5.89	1.94
6	1.53	97.75	49.43	75.45	3.04	27.62	224.32	6.11	58.93	34.96	Sandy silt	Grey	7.99	1.20
7	1.56	85.38	46.06	72.03	2.30	10.45	78.25	9.26	76.85	13.89	Sandy silt	Grey	7.28	1.49
8	1.69	60.76	37.79	63.82	2.51	15.15	160.85	8.16	65.95	25.89	Sandy silt	Dark grey	3.60	0.55
9	1.67	70.52	41.35	68.87	2.33	10.26	63.27	9.10	80.05	10.85	Sandy silt	Dark grey	5.74	6.52
10	1.58	92.01	47.92	75.80	2.58	13.57	779.07	7.85	77.04	15.11	Sandy silt	Greyish brown	5.95	24.30
11	1.49	108.66	52.07	77.81	2.39	9.98	55.33	8.70	82.62	8.68	Sandy silt	Greyish brown	7.55	1.83
13	1.56	82.79	45.29	70.61	2.50	12.75	110.08	8.17	72.79	19.04	Silt	Greyish brown	6.20	2.14
14	1.72	49.70	33.20	56.96	3.30	43.54	135.85	6.19	55.25	38.56	Sandy silt	Greyish brown	6.18	1.59
15	1.64	68.04	40.49	66.25	2.74	16.63	97.66	7.08	72.27	20.65	Sandy silt	Greyish brown	5.16	0.14
16	1.63	66.17	39.82	64.91	2.92	28.09	108.01	6.54	66.25	27.21	Sandy silt	Greyish brown	4.98	0.98
17	1.58	77.92	43.79	69.14	2.69	15.98	80.94	7.38	75.94	16.68	Sandy silt	Greyish brown	3.03	1.17
18	1.52	76.92	43.48	66.04	2.55	12.88	71.55	7.93	78.57	13.50	Sandy silt	Greyish brown	4.99	0.43
19	1.36	125.60	55.67	75.56	2.43	10.08	57.94	8.43	81.57	9.82	Silt	Grey	7.19	0.59
20	1.80	41.83	29.49	52.97	2.25	10.70	183.95	9.57	64.04	26.39	Sandy silt	Greyish brown	2.60	0.76
21	1.61	71.11	41.56	67.06	2.91	21.03	114.42	6.60	70.19	23.21	Sandy silt	Greyish brown	4.55	0.40
22	1.60	77.10	43.53	69.74	3.04	40.51	212.00	6.58	52.90	40.52	Sandy silt	Greyish brown	4.57	0.83
23	1.68	54.10	35.11	58.81	2.94	24.20	175.63	6.40	62.55	31.05	Sandy silt	Greyish brown	4.24	0.73
24	1.62	65.85	39.70	64.44	2.92	26.97	121.43	6.52	63.61	29.87	Sandy silt	Greyish brown	4.23	0.99
25	1.54	89.64	47.27	72.59	3.02	30.50	172.72	6.20	60.27	33.53	Sandy silt	Greyish brown	3.97	0.92
27	1.85	38.70	27.90	51.50	2.03	9.01	158.75	11.18	68.36	20.46	Sandy silt	Grey	2.31	0.64
28	1.67	62.68	38.53	64.37	3.07	45.18	247.27	6.56	48.92	44.52	Sandy silt	Grey	4.40	1.02
29	1.76	53.40	34.81	61.12	3.29	69.98	293.96	5.98	41.74	52.28	Silty sand	Grey	3.36	0.74
30	1.83	41.14	29.15	53.27	3.91	134.47	318.08	4.85	28.86	66.29	Silty sand	Grey	3.05	1.02
31	1.77	49.03	32.90	58.27	2.91	48.72	283.00	6.82	46.05	47.13	Sandy silt	Grey	3.33	0.81
32	1.61	76.97	43.49	70.10	2.17	11.29	92.09	10.12	70.60	19.28	Sandy silt	Greyish brown	3.79	0.46
33	1.72	65.50	39.58	68.07	2.77	30.79	147.44	6.96	55.27	37.77	Sandy silt	Greyish brown	5.03	0.74
34	1.53	79.48	44.28	67.76	3.38	165.34	391.17	5.78	35.02	59.20	Silty sand	Greyish brown	5.45	0.82
35	1.93	32.64	24.61	47.56	3.81	204.28	488.65	4.80	34.38	60.82	Silty sand	Greyish brown	1.75	0.88

Table 2-5. General characteristics of sediment collected from *lower* surface sediments within Coombabah Lake (continued)

Sample grid	ρ_{wb} (g cm^{-3})	Θ_D (%)	Θ_W (%)	ϕ (%)	Sediment grain size distribution (µm)			Wentworth size class (%)			Texture	Colour	LOI_{550} (%)	LOI_{950} (%)
					$d_{(0.1)}$	$d_{(0.5)}$	$d_{(0.9)}$	Clay	Silt	Sand				
36	1.85	40.12	28.63	53.08	4.01	167.62	421.87	4.72	30.26	65.02	Silty sand	Greyish brown	2.78	0.95
37	1.81	45.47	31.26	56.65	3.23	73.89	320.40	5.14	42.17	52.69	Silty sand	Greyish brown	2.90	0.54
38	1.86	39.20	28.16	52.43	3.61	92.95	283.66	12.00	30.35	57.65	Silty sand	Greyish brown	2.77	0.35
39	1.82	44.81	30.94	56.20	7.78	156.55	259.51	6.18	10.57	83.25	Silty sand	Greyish brown	2.20	0.45
40	1.94	38.05	27.56	53.54	7.54	197.54	379.16	6.20	15.02	78.78	Silty sand	Greyish brown	2.02	0.53
41	1.80	39.41	28.27	50.90	8.26	196.66	352.43	5.75	12.09	82.16	Silty sand	Greyish brown	1.39	0.46
42	2.01	28.52	22.19	44.57	6.03	247.40	491.84	7.20	18.94	73.86	Silty sand	Greyish brown	1.62	0.58
43	1.61	72.27	41.95	67.66	3.39	45.17	213.48	13.34	42.34	44.32	Sandy silt	Greyish brown	5.98	0.76
44	1.75	57.31	36.43	63.82	3.04	54.23	156.36	14.83	38.50	46.67	Sandy silt	Olive grey	3.84	0.62
45	1.76	48.58	32.69	57.66	5.12	159.36	329.05	8.53	21.15	70.32	Silty sand	Olive grey	3.31	0.80
46	1.93	31.07	23.70	45.79	22.96	207.44	324.31	4.11	7.68	88.21	Silty sand	Light olive grey	1.43	0.41
47	1.71	59.52	37.31	63.91	3.63	155.57	358.60	12.07	23.06	64.87	Silty sand	Greyish brown	4.87	0.78
48	1.71	54.46	35.26	60.46	4.03	58.27	316.03	10.94	39.54	49.52	Sandy silt	Greyish brown	8.19	0.89
49	1.81	45.39	31.22	56.40	3.55	40.87	224.75	12.48	49.26	38.26	Sandy silt	Greyish brown	3.29	0.41
50	2.05	28.64	22.27	45.59	115.21	225.72	370.82	2.95	5.41	91.64	Sand	Light grey	1.31	0.41
51	1.94	25.00	20.00	38.89	165.40	256.31	396.37	0.00	0.00	100.00	Sand	Light grey	0.47	0.62
52	1.61	62.57	38.49	61.85	3.53	31.25	223.00	12.99	91.64	35.39	Sandy silt	Grey	5.40	0.75
54	1.88	38.77	27.94	52.56	5.73	202.09	374.09	3.34	20.29	76.37	Silty sand	Grey	1.86	0.94
55	1.84	41.11	29.13	53.69	2.76	133.50	377.69	7.25	43.53	49.22	Sandy silt	Grey	3.37	0.37
56	1.53	97.20	49.29	75.41	2.77	12.90	79.72	6.84	78.77	14.39	Silty sand	Greyish brown	7.86	0.66
58	1.79	47.54	32.22	57.60	5.63	216.53	355.61	2.77	19.49	77.74	Silty sand	Grey	4.02	0.55
59	1.86	40.42	28.78	53.59	4.93	178.15	345.70	3.67	19.90	76.43	Silty sand	Grey	2.37	0.37

Grain size distribution

Grain size distributions and classes varied within the lake system, although sandy-silts and silty-sands dominate the surface sediments of Coombabah Lake. The median particle size, $d_{(0.5)}$, at both the upper and lower sample depths showed a tendency for the <63 µm fraction to dominate the freshwater input end of the lake system and the >63 µm fraction to generally dominate the marine end of the lake (Figure 2-3.). Higher currents observed at the marine end of the lake as a result of tidal forcing and the input of fluvial sediments from the catchment into the freshwater end are believed to influence this pattern, with coarser sediments predominating at the northern entrance. Differences between the sampled depths are patchy and do not show general patterns.

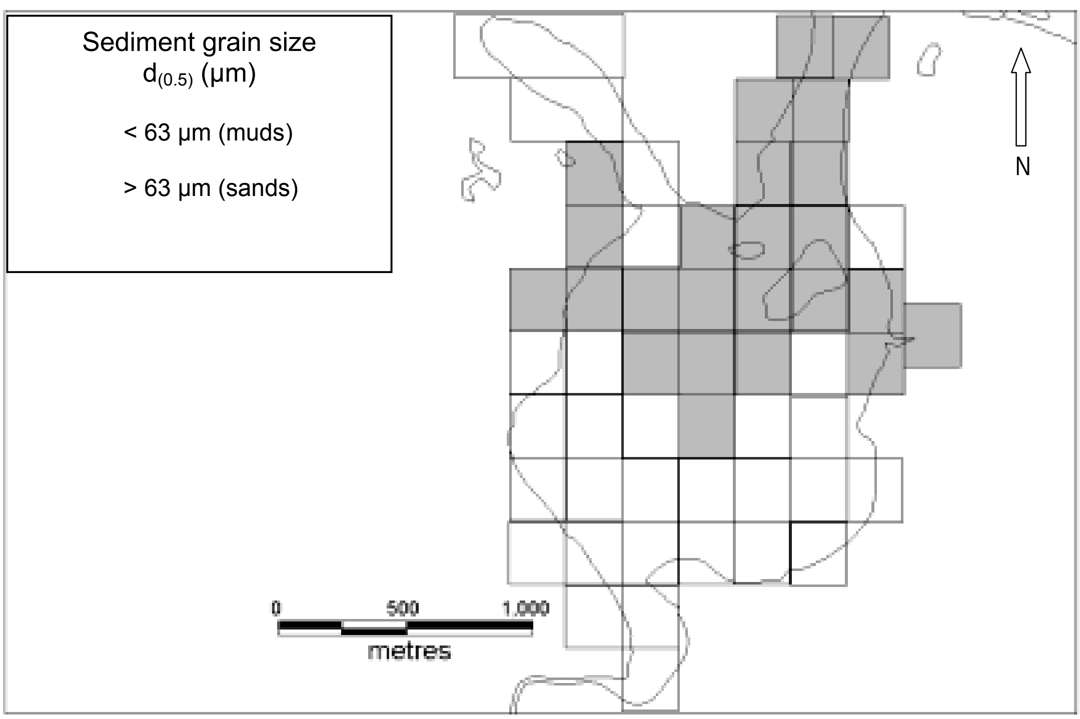

Figure 2-3. Mud- and sand-dominated sediment distribution of Coombabah Lake

The temporal variability of grain size distributions in estuarine sediment has been demonstrated but conclusions as to dominant processes controlling those changes are varied (Penthick 1984). Changes in tidal velocity and wind speed (both with daily variability superimposed upon a seasonal pattern of change) play an important role in resuspending bed sediment and thus could greatly influence the proportion of fine particles in the surface sediment. Under high discharge conditions, rivers also remove fine particles from surface bottom sediment (Penthick 1984). Particles are transformed by processes such as decomposition, mineralisation, dissolution, adsorption, coagulation, precipitation, faecal pellet production and bacterial degradation.

Organic matter and carbonate content

The fate of organic carbon in the estuarine environment is determined by a number of factors including the allochthonous import of material from terrigenous and marine reservoirs, internal productivity and recycling (Ruddy 1997; Alongi *et al*. 1999). In most marine-type sediments organic carbon is the only reducing agent to enter a sediment column. The remainder of the sediment load arrives in its oxidised form, and with the exception of straightforward compaction, early diagenesis (i.e. the process of change during burial) results directly or indirectly from the flow of electrons (Kristensen & Holmer 2001). Of the organic matter deposited on the sediment surface, 30–99% is remineralised rapidly during early diagenesis (Ruddy 1997). The rate at which organic matter is supplied to estuaries determines the trophic status of the water body (Eyre & Ferguson 2002). The trophic status of an estuary should not be confused with the term eutrophication, which is used to describe an increase in the rate of supply of organic matter into the system.

Sediment organic matter content values, LOI_{550}, within Coombabah Lake (Figure 2-4) were primarily greatest at sample grids in the southern areas of the lake at both the upper and lower surface sediment depths (Tables 2-4 and 2-5). Decreases in sediment grain size coincided with increases in organic matter content. The accumulation of the fine sediments and organic matter in the southern sediments can be attributed to inputs from catchment/urban sources into the shallow lake. Sediment LOI_{950} values were patchy throughout the system. Sediments exhibited similar LOI_{550} and LOI_{950} values at both the upper and lower surface depths respectively.

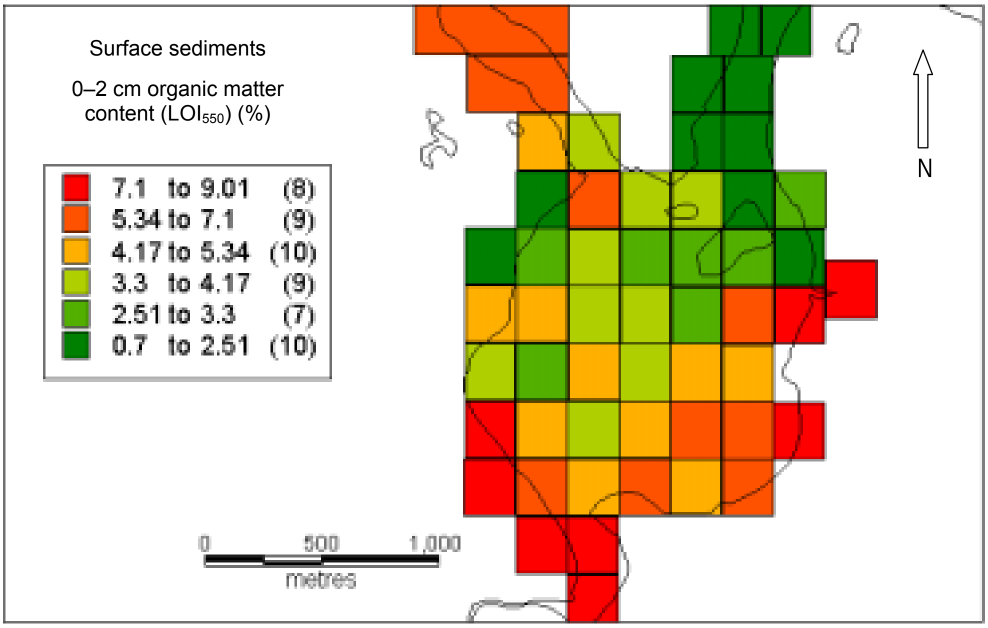

Figure 2-4. Organic matter content within 0–2 cm depth surface sediments of Coombabah Lake

Sediment colour

Sediment colours, determined using the Munsell® colour chart, varied only to a small extent. Samples were dominated by the 2.5Y hue. Grey, dark grey, greyish-brown, olive grey and light grey colours dominated the samples and all samples were characterised by the 2.5Y hue, similar to that of the 0–2 cm samples (Tables 2-4 and 2-5).

Chlorophyll *a* and phaeopigment concentrations

Present and past seagrass and phytoplankton surveys indicate high numbers of phytoplankton, including toxic red algae (*Heterosigma akashiwo*) (Waltham *et al.* 2002) and a total lack of seagrass on the lake floor (Frank & Fielding 2004). Surface sediment chl *a* and phaeo concentrations were patchy and showed no distinctive spatial patterns (Figure 2-5).

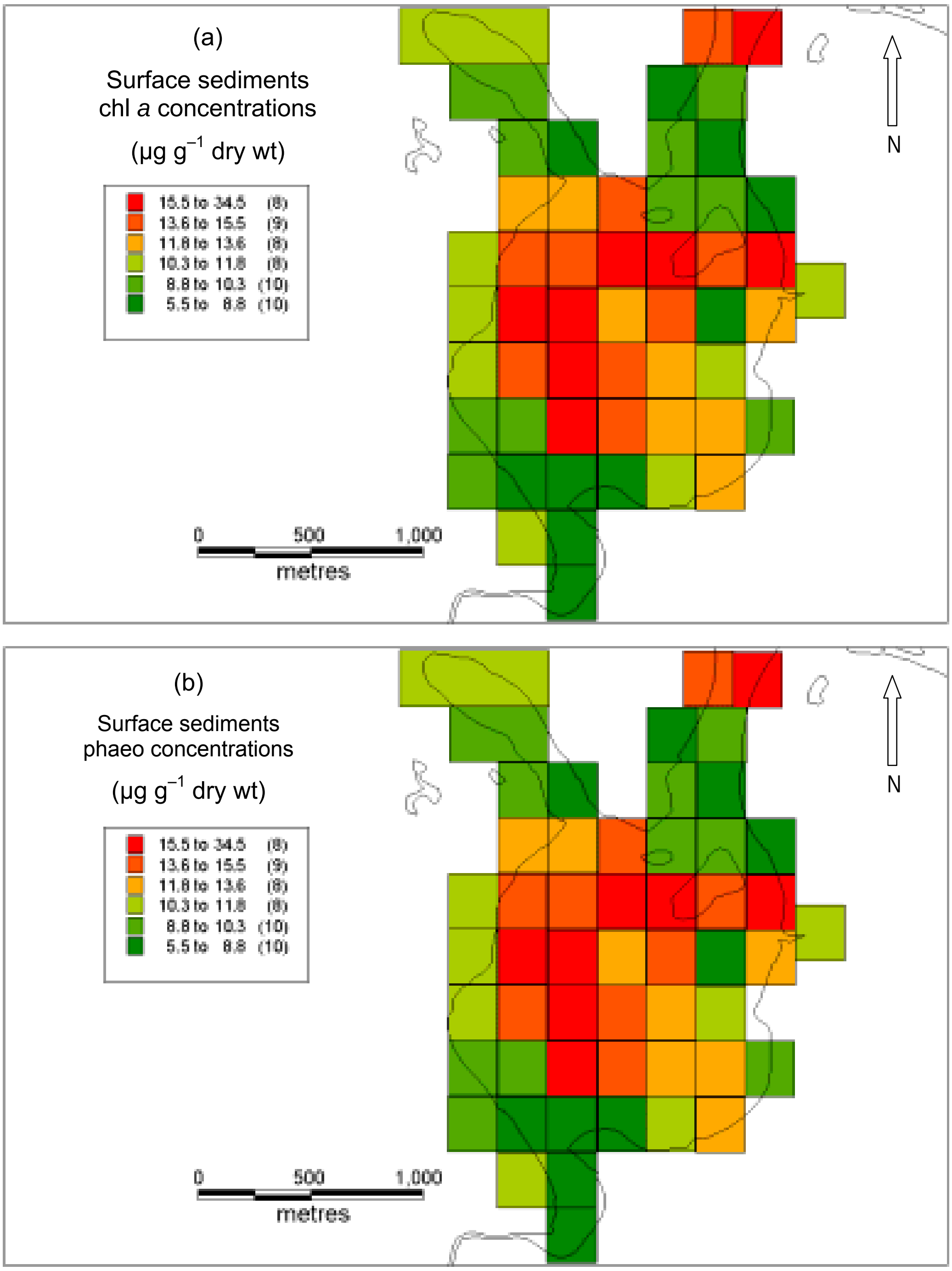

Figure 2-5. Concentrations of (a) chlorophyll *a* and (b) phaeopigments within 0–2 cm depth surface sediments of Coombabah Lake

Mineralogy

The input of sediment to coastal systems originates from marine, coastal, fluvial and *in situ* sources including solid materials biologically and chemically precipitated from waters within the depositional basin, erosion of upland bluffs, terrestrial runoff, aeolian transport and transport from ocean inlets, estuarine mouths and deeper portions of estuaries (Eisma 1992). Sediments, however, consist primarily of mineral grains originating from weathering and erosion of pre-existing rock masses in addition to both fluvial and offshore transported sediments to the depositional site. Shells and body parts from estuarine organisms also make an important contribution to the sediment of the estuary (Davis Jr. 1997).

Depending on the local situation, one or more sources will dominate the supply. The dominant source may be observed from the composition of the suspended sediment ratios from both sources. Sediment supplied from the sea contains particles produced by marine organisms (e.g. diatom frustles, sea urchin spines, and shell fragments), which serve as tracers of marine origin (Davis Jr. 1997). Finer sands and muds within estuarine systems generally originate from fluvial sources, as opposed to coarser sands derived from the shelf or from erosion of the shoreline. Surficial sedimentary deposits reflect the interaction between fluvial processes, which decrease in strength seaward, and marine processes, which decrease in strength landwards.

Sediment mineralogy within Coombabah Lake is characterised by halite, pyrite, anorthite, quartz and kaolinite which predominate in mixed groupings within the upper and lower sampled sediments (Table 2-6). Pyrite and kaolinite in both sample depths were observed in samples characterised by small grain sizes and occurring within the freshwater entrance region of the lake. At the northern end of the lake sediments comprise quartz and halite (Table 2-6.)

Nutrients within surface sediments of the lake

The measured concentrations of nutrients in the 0–2 cm (upper) and 2–5 cm (lower) surface sediments from Coombabah Lake are summarised in Table 2-7. Surface sediments within Coombabah Lake were characterised by mean values of 0.85 ± 0.42 % organic C and 0.06 ± 0.03 % organic N. Higher % organic C and N values were observed for sediments with finer particle sizes.

Table 2-6. Surface sediment mineralogy of Coombabah Lake

Sample grid	Mineralogy of upper (0–2 cm) surface sediment	Mineralogy of lower (2–5 cm) surface sediments
1	Halite, pyrite, anorthite, quartz and kaolinite	Halite, pyrite, anorthite, quartz and kaolinite
3	Halite, anorthite, quartz and kaolinite	Halite, pyrite, anorthite, quartz and kaolinite
4	Halite, pyrite, anorthite, quartz and kaolinite	Halite, pyrite, anorthite, quartz and kaolinite
6	Halite, pyrite, anorthite, quartz and kaolinite	Halite, pyrite, anorthite, quartz and kaolinite
7	Halite, anorthite, quartz and kaolinite	Halite, pyrite, anorthite, quartz and kaolinite
8	Halite, anorthite and quartz	Halite, pyrite, anorthite, quartz and kaolinite
9	Halite, pyrite, anorthite, quartz and kaolinite	Halite, pyrite, anorthite, quartz and kaolinite
10	Halite, pyrite, anorthite, quartz and kaolinite	Halite, anorthite, quartz and kaolinite
11	Halite, pyrite, anorthite, quartz and kaolinite	Halite, anorthite, quartz and kaolinite
13	Halite, anorthite, quartz and kaolinite	Halite, anorthite, quartz and kaolinite
14	Halite, pyrite, anorthite and quartz	Halite, anorthite and quartz
15	Halite, anorthite, quartz and kaolinite	Halite, anorthite and quartz
16	Halite, anorthite, quartz and kaolinite	Halite, anorthite and quartz
17	Halite, pyrite, anorthite, quartz and kaolinite	Halite, anorthite, quartz and kaolinite
18	Halite, pyrite, anorthite, quartz and kaolinite	Halite, anorthite, quartz and kaolinite
19	Halite, anorthite, quartz and kaolinite	Halite, anorthite, quartz and kaolinite
20	Halite, anorthite and quartz	Halite, anorthite and quartz
21	Anorthite and quartz	Halite, anorthite, quartz and kaolinite
22	Halite, anorthite, quartz and kaolinite	Halite, anorthite, quartz and kaolinite
23	Halite, anorthite, quartz and kaolinite	Halite, anorthite, quartz and kaolinite
24	Halite, anorthite, quartz and kaolinite	Halite, anorthite, quartz and kaolinite
25	Halite, anorthite, quartz and kaolinite	Halite, anorthite, quartz and kaolinite
27	Halite, anorthite, quartz and kaolinite	Halite, quartz and kaolinite
28	Halite, anorthite, quartz and kaolinite	Halite, anorthite and quartz
29	Halite, anorthite, quartz and kaolinite	Halite, anorthite and quartz
30	Halite, anorthite, quartz and kaolinite	Halite, anorthite, quartz and kaolinite
31	Halite, anorthite and quartz	Halite, anorthite and quartz
32	Halite, anorthite, quartz and kaolinite	Halite, pyrite, anorthite, quartz and kaolinite
33	Halite, anorthite, quartz and kaolinite	Halite, anorthite, quartz and kaolinite
34	Halite, anorthite, quartz and kaolinite	Halite, anorthite and quartz
35	Halite and quartz	Halite, anorthite and quartz
36	Halite, anorthite and quartz	Halite, anorthite and quartz
37	Halite, anorthite, quartz and kaolinite	Halite, anorthite and quartz
38	Halite, anorthite, quartz and kaolinite	Halite, anorthite and quartz
39	Halite, anorthite and quartz	Halite, anorthite and quartz
40	Halite, anorthite and quartz	Halite and quartz
41	Halite, anorthite and quartz	Halite and quartz
42	Halite, anorthite and quartz	Halite and quartz
43	Halite, anorthite, quartz and kaolinite	Halite, anorthite, quartz and kaolinite
44	Halite, anorthite, quartz and kaolinite	Halite, anorthite and quartz
45	Halite, anorthite and quartz	Halite, anorthite and quartz
46	Halite and quartz	Halite and quartz
47	Halite, pyrite, anorthite and quartz	Halite, anorthite and quartz
48	Halite, anorthite and quartz	Halite, anorthite and quartz
49	Halite, anorthite and quartz	Halite, anorthite and quartz
50	Halite and quartz	Halite and quartz
51	Halite and quartz	Halite and quartz
52	Halite, anorthite, quartz and kaolinite	Halite, anorthite, quartz and kaolinite
54	Halite and quartz	Halite, anorthite and quartz
55	Halite, anorthite, quartz and kaolinite	Halite and quartz
56	Halite, anorthite, quartz and kaolinite	Halite and quartz
58	Halite and quartz	Halite and quartz
59	Halite and quartz	Halite and quartz

Table 2-7. Nutrient concentrations of sediments collected from 0–2 cm (upper) and 2–5 cm (lower) sediments of sample grids within Coombabah Lake

Grid	% C	$\delta^{13}C$ (‰)	% N	$\delta^{15}N$ (‰)	Total phosphorus ($\mu g\ g^{-1}$ dry wt)		1M $MgCl_2$ extractable P ($\mu g\ g^{-1}$ dry wt)		1M KCl extractable NH_4^+ ($\mu g\ g^{-1}$ dry wt)	
	0–2 cm	0–2 cm	0–2 cm	0–2 cm	0–2 cm	2–5 cm	0–2 cm	2–5 cm	0–2 cm	2–5 cm
1	–		–	–	355.15	497.11	0.97	1.06	13.18	25.98
3	–		–	–	292.07	447.28	1.28	1.23	11.38	22.35
4	1.76	–25.3	0.10	2.8	998.25	443.92	0.87	0.95	13.45	25.57
6	1.58	–24.5	0.10	2.8	401.85	420.48	0.65	1.01	10.97	19.98
7	1.29	–24.4	0.09	2.8	323.80	400.18	0.43	0.55	5.21	10.62
8	1.45	–24.9	0.07	3.3	310.38	469.53	1.03	0.67	2.23	3.04
9	1.35	–23.5	0.07	2.9	350.82	405.61	0.36	0.76	1.84	3.00
10	1.12	–23.8	0.08	3.3	313.58	593.03	0.41	0.50	2.05	3.52
11	1.23	–23.5	0.10	3.5	316.13	494.31	1.39	0.83	2.63	4.54
13	1.07	–23.3	0.09	3.3	289.15	713.64	0.53	0.64	9.19	16.26
14	0.87	–25.1	0.05	1.7	266.89	418.60	0.21	0.48	4.14	5.74
15	0.88	–24.1	0.06	2.3	256.31	552.25	0.35	0.38	4.53	8.49
16	0.84	–23.8	0.06	2.0	310.23	571.29	0.32	0.58	2.90	5.88
17	1.00	–23.6	0.07	2.3	362.75	514.04	0.49	0.53	2.09	3.71
18	1.06	–23.7	0.08	2.3	400.38	534.30	0.58	0.55	1.98	6.83
19	1.25	–23.5	0.10	2.7	391.50	335.04	0.48	0.82	2.07	5.07
20	0.84	–24.3	0.06	2.4	236.09	250.36	0.23	0.25	2.04	2.76
21	0.76	–23.9	0.06	2.0	281.86	452.21	0.32	0.32	2.32	4.48
22	0.70	–23.3	0.05	2.0	350.65	435.34	0.18	0.38	2.21	4.28
23	0.66	–23.7	0.05	2.3	137.30	435.99	0.23	0.17	1.61	2.66
21	1.33	–24.6	0.08	2.0	296.84	428.73	0.25	0.28	1.60	3.78
25	0.88	–23.5	0.07	2.0	375.25	389.23	0.23	0.37	2.01	4.15
27	0.88	–23.0	0.08	2.4	296.92	291.12	0.32	0.22	3.27	4.41
28	–	–	–	–	276.27	395.15	0.27	0.28	2.61	4.07
29	0.63	–23.3	0.05	2.2	205.95	353.65	0.27	0.26	2.21	3.27
30	0.63	–23.4	0.05	2.9	226.49	497.11	0.27	0.21	0.91	1.26
31	0.47	–23.3	0.04	2.5	323.32	447.28	0.34	0.16	0.73	1.11
32	1.01	–23.3	0.07	2.3	243.27	443.92	0.33	0.24	0.85	1.44
33	1.05	–24.2	0.07	3.0	326.67	420.48	0.31	0.21	2.48	3.91
34	1.31	–24.3	0.09	3.0	364.25	350.09	0.54	0.30	3.64	6.08

Table 2-7. Nutrient concentrations of sediments collected from 0–2 cm (upper) and 2–5 cm (lower) sediments of sample grids within Coombabah Lake (continued)

Grid	% C	$\delta^{13}C$ (‰)	% N	$\delta^{15}N$ (‰)	Total phosphorus (µg g–l dry wt)		1M MgCl2 extractable P (µg g–l dry wt)		1M KCl extractable NH4+ (µg g–l dry wt)	
	0–2 cm	0–2 cm	0–2 cm	0–2 cm	0–2 cm	2–5 cm	0–2 cm	2–5 cm	0–2 cm	2–5 cm
35	0.33	–23.4	0.03	2.1	98.48	127.32	0.17	0.18	0.72	0.92
36	0.46	–22.7	0.04	3.0	217.89	260.18	0.19	0.20	1.10	1.54
37	0.50	–23.0	0.04	2.4	234.53	369.29	0.19	0.16	1.17	1.67
38	–	–	–	–	185.95	326.99	0.29	0.14	1.00	1.50
39	0.54	–25.0	0.03	3.4	62.05	226.04	0.14	0.13	1.09	1.62
40	0.37	–22.8	0.03	3.3	133.95	281.44	0.16	0.08	1.59	1.44
41	0.35	–23.5	0.03	4.1	137.57	253.46	0.20	0.14	0.87	1.19
42	0.34	–22.8	0.03	2.2	223.14	184.55	0.16	0.11	0.47	0.62
43	1.30	–25.2	0.08	3.1	335.32	337.28	0.22	0.15	0.63	1.10
44	0.77	–24.2	0.05	2.3	261.86	415.25	0.27	0.29	0.94	1.31
45	0.64	–23.5	0.05	2.6	257.86	281.30	0.23	0.28	0.91	1.30
46	0.13	–24.6	0.01	4.0	73.75	165.76	0.15	0.15	1.16	1.40
47	0.87	–20.9	0.06	3.4	201.03	399.30	0.21	0.11	0.88	1.29
48	1.10	–25.8	0.06	1.7	216.32	327.32	0.27	0.18	2.81	4.75
49	0.58	–23.6	0.04	2.4	279.62	396.83	0.22	0.35	0.88	1.17
50	0.22	–22.9	0.02	4.6	5.59	98.48	0.09	0.24	1.22	1.27
51	< MDL	–	0.01	4.8	1.68	103.86	0.11	0.29	0.87	0.86
52	1.52	–26.1	0.08	2.5	250.11	361.07	0.65	0.67	8.26	15.18
54	0.22	–23.8	0.02	1.8	33.61	202.80	0.14	0.28	0.42	0.46
55	0.75	–23.3	0.07	4.2	194.52	282.00	0.21	0.32	0.64	0.57
56	1.65	–25.4	0.12	2.6	399.50	537.14	0.58	1.09	9.57	19.42
58	0.12	–25.4	0.01	4.8	1.68	177.75	0.08	0.32	0.43	0.70
59	0.32	–24.0	0.03	3.8	149.17	170.87	0.14	0.29	0.72	0.81

– represents no sample analysis
< MDL represents below method detection limit

Results from the selected nutrient analyses for the upper and lower surface sediments ranged from 0.12% to 1.76% for % organic C and 0.01 % to 0.12% for % organic N, 1.68 μg g^{-1} to 998.25 μg^{-1} for TP as phosphorous, 0.08 μg g^{-1} to 1.39 μg g^{-1} for E-P as phosphorous and 0.42 μg g^{-1} to 26.41 μg g^{-1} for E-NH_4^+. Variability (RSD) within the lake floor upper surface sediments included: 49.72% (% organic C), 45.37% (% organic N), 56.90% (TP), 78.22% (E-P), 113.32% (E-NH_4^+) respectively and 35.65% (TP), 60.43 % (E-P) and 113.40 % (E-NH_4^+) within the lower surface sediments.

Mean nutrient concentrations were on average greater in the lower surface sediments (369.50 ± 131.27 μg g^{-1}) compared to the upper surface sediments (261.17 ±148.60 μg g^{-1}) for TP ,E-NH_4^+ (5.16 ± 6.81 μg g^{-1}, 2–5 cm; and 2.96 ± 3.35 μg g^{-1}, 0–2 cm) and E-P (0.40 ± 0.29 μg g^{-1}, 2–5 cm; and 0.37± 0.29 μg g^{-1}, 0–2 cm). Paired t-tests (2-tailed) indicated significant differences between the observed upper and lower surface sediment concentrations for TP ($\alpha = 0.05$, $df = 52$, $t = -6.097$, $p < 0.001$) and E-NH_4^+ ($\alpha = 0.05$, $df = 52$, $t = -5.190$, $p < 0.001$). A non-significant difference between the upper and lower surface sediments for E-P concentrations ($\alpha = 0.05$, $df = 52$, $t = -1.401$, $p = 0.167$) was observed.

Pearson correlations demonstrated significant correlations between all surface sediment nutrients analysed (Figure 2-6).

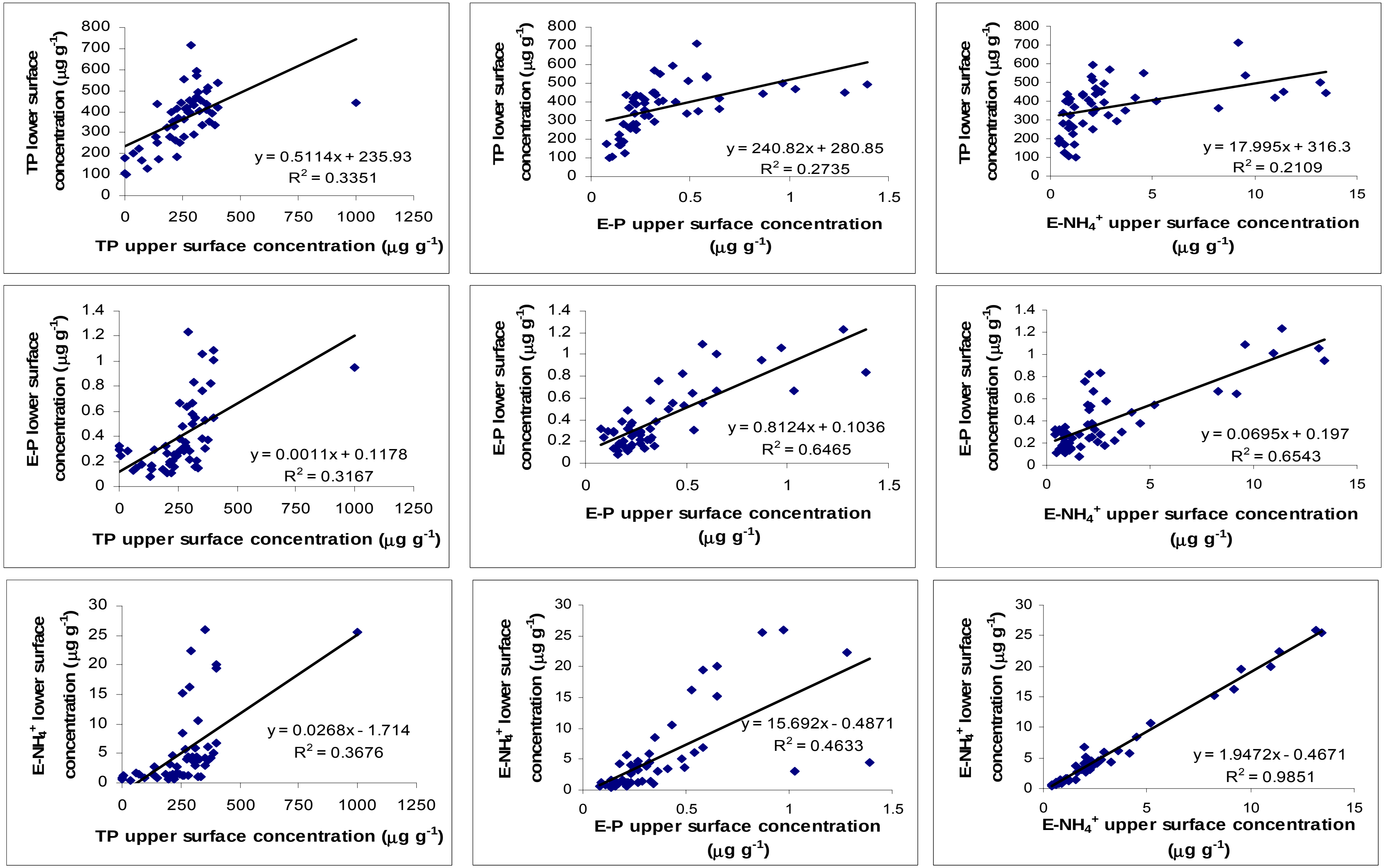

Figure 2-6. Correlation matrix for Coombabah Lake surface sediment concentrations (upper and lower surface depths)

Significant correlations were observed between the upper and lower surface sediment concentrations for TP, E-P and E-NH_4^+ and between each of the selected nutrients at each surface depth (Table 2-8).

Table 2-8. Pearson correlation between upper (0–2 cm) and lower (2–5 cm) surface sediment nutrient concentrations (n = 53)

Surface sediment nutrient concentrations		TP (0–2 cm)	E-P (0–2 cm)	E-NH_4^+ (0–2 cm)
TP (2–5 cm)	Pearson correlation	0.579**	0.523**	0.459**
	Sig. (2-tailed)	0.000	0.000	0.000
E-P (2–5 cm)	Pearson correlation	0.563**	0.804**	0.809**
	Sig. (2-tailed)	0.000	0.000	0.000
E-NH_4^+ (2–5 cm)	Pearson correlation	0.606**	0.681**	0.993**
	Sig. (2-tailed)	0.000	0.000	0.000

** Correlation is significant at the 0.01 level

Significant correlations were also observed for all nutrient concentrations and median grain sizes ($d_{(0.5)}$) within Coombabah Lake at both the upper and lower surface sediment depths (Table 2-9).

Table 2-9. Pearson correlation between upper (0–2 cm) and lower (2–5 cm) surface sediment median grain sizes ($d_{(0.5)}$) and selected nutrient concentrations (n = 53)

Upper surface sediments		TP	E-P	E-NH4+
$d_{(0.5)}$	Pearson correlation	–0.637**	–0.546**	–0.498**
	Sig. (2-tailed)	0.000	0.000	0.000
Lower surface sediments		**TP**	**E-P**	**E-NH4+**
$d_{(0.5)}$	Pearson correlation	–0.804**	–0.540**	–0.476**
	Sig. (2-tailed)	0.000	0.000	0.000

** Correlation is significant at the 0.01 level

In general, low nutrient concentrations were recorded for TP, E-P and E-NH_4^+ in the upper and lower surface sediments in sample grids dominated by low mud contents, most noticeably within the northern lake entrance. Increased concentrations for all nutrients analysed were observed in both depths dominated by high mud content, occurring predominantly in the southern region of the lake. Average concentrations in sediments from the southern sample grids (1–25) were greater for all nutrients measured compared to sediments from the northern sandier half of the lake grids (27–59) (Table 2-10), where sediments originate from the coastal zone. Sampled grids within the northern half of the lake in the northwest grids (43, 49, 52 and 56) are mud-dominated.

Table 2-10. Average sediment nutrient concentrations from 0–2cm and 2–5 cm sediments within the southern and northern regions of Coombabah Lake

Nutrient	Depth	Sample grid localities	
		South (grids 1–25)	North (grids 27–56)
% organic C	0–2 cm	1.10 ± 0.30%	0.68 ± 0.41%
% organic N	0–2 cm	0.07 ± 0.02%	0.05 ± 0.03%
TP	0–2 cm	346 ± 157.73 µg g^{-1}	200.81 ± 108.58 µg g^{-1}
	2–5 cm	463.75 ± 95.30 µg g^{-1}	302.62 ± 111.24 µg g^{-1}
E-P	0–2 cm	0.54 ± 0.35 µg g^{-1}	0.25 ± 0.13 µg g^{-1}
	2–5 cm	0.61 ± 0.28 µg g^{-1}	0.26 ± 0.19 µg g^{-1}
E-NH_4^+	0–2 cm	4.62 ± 4.07 µg g^{-1}	1.78 ± 2.09 µg g^{-1}
	2–5 cm	8.76 ± 7.80 µg g^{-1}	2.83 ± 4.14 µg g^{-1}

Carbon:nitrogen ratios (C:N) within the lake sediments ranged between 9.00 and 20.18. C:N values were greatest in sample grids 4, 8, 9, 14, 38, 48 and 52 (>18.30). Sample grids 4, 8, 9 and 14 were grouped at the entrance of Coombabah Creek at the freshwater end and sample grids 48 and 52 were connected adjacent to the lakeshore in the northwest arm. Both of these areas are dominated by high-density mangrove environments, which would generate high C:N detritus. The mean C:N values for the lake of 13.94 ± 2.69 compare well with the Australian coastal and estuarine sediments C:N average of 10.1–24.3 (Heap *et al.* 2001).

The δ^{13}C values of the surface sediments ranged from -20.9 to -26.1‰ with an average of -23.9 ± 0.9‰. Depleted δ^{13}C values (-26.1 to -25.2‰) are grouped in the northwest arm of the lake (grids 43, 48, 52 and 58), in addition to a -25.3‰ value located at the freshwater entrance of Coombabah creek (grid 4). The δ^{15}N values ranged from 1.7–4.8‰ with an average value of 2.8 ± 0.8‰. Enriched δ^{15}N values were predominantly found at the entrances of the lake at both the marine and freshwater ends.

Measured δ^{13}C values of the upper sediments were similar and within the range reported for mangroves of -24 to -30 ‰ (Hemminga *et al*. 1994; Qian *et al*. 1996; Marguillier *et al*. 1997). Additionally δ^{15}N values corresponded to previous studies of values approximating and below +2 ‰ for mangroves (Boutton 1991; Coffin *et al.* 1994; Nadelhoffer & Fry 1994); however a large proportion of the sampled sediments indicate enriched δ^{15}N values (> +2 ‰).

Nutrient variations and inputs to the lake environment

Differences in nutrient concentrations are attributable to variations in sediment source, grain size, NPS catchment runoff, hydrology and the distance to specific urban development–related (i.e. stormwater runoff) nutrient sources (point sources).

Increased TP concentrations both at upper and lower surface depths occurred in sediments: (i) within the freshwater entrance (sample grids 1–8); (ii) adjacent to the golf course development (sample grids 9, 10 and 17); (iii) opposite the sewage treatment plant (sample grids 18, 19 and 25); (iv) within the small creek entering the lake system on the eastern lake side (sample grids 33 and 34); and (v) in sample grids 22 and 56.

Data exists for surface sediment TP concentrations within coastal and estuarine systems in Australia (Heap *et al*. 2001), which allows for direct comparisons with the data obtained from Coombabah Lake. On average, TP concentrations observed within the lake compared well with concentrations typically encountered within Australian coastal and estuarine systems (Figure 2-7). TP concentrations, when compared to the Australian average of 120–450 $\mu g\ g^{-1}$ (Heap *et al*. 2001), indicate typical estuarine sediment TP concentrations; however, subsurface samples within the south region of the lake are on average greater than 450 $\mu g\ g^{-1}$ ($463.75 \pm 95.30\ \mu g\ g^{-1}$).

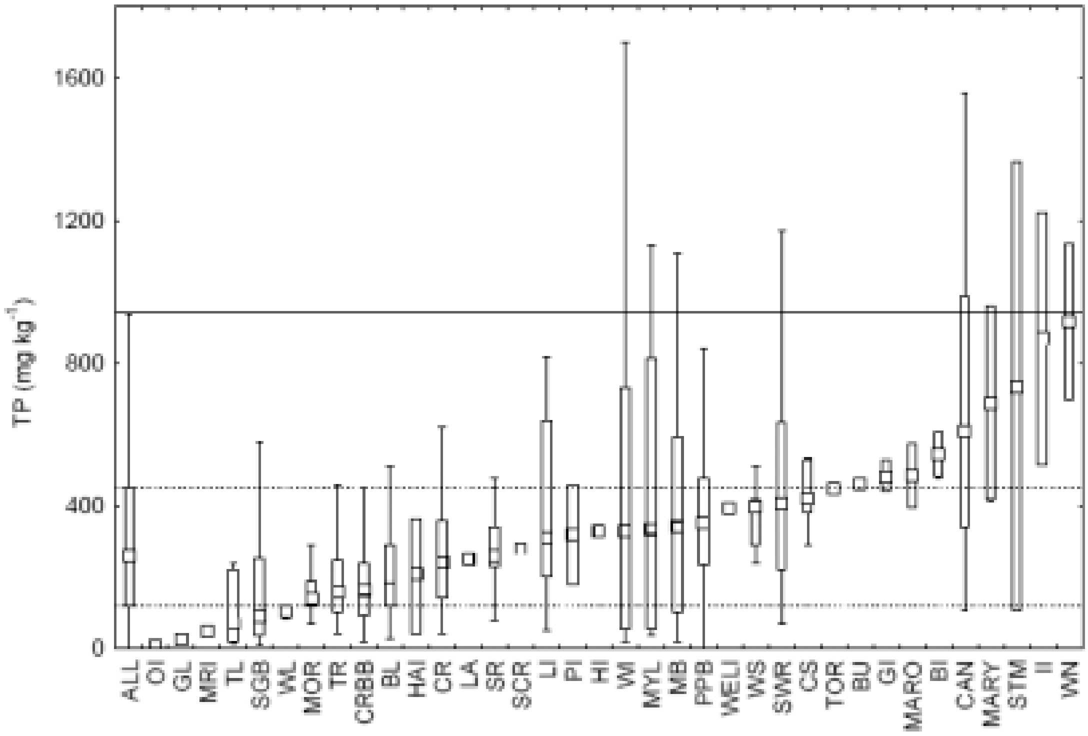

Figure 2-7. Total phosphorous sediment concentrations from Australian estuaries and waterways

The dotted lines mark the non-outlier 25th and 75th percentiles respectively of Australian coastal and estuarine systems, and include the range within which 50% of the data lies. The solid line marks the non-outlier maximum of the total data set. Values found above these lines are extreme. Abbreviated names are as follows:

ALL: All systems
BI: Beaufort Inlet, Western Australia
BU: Burnett River, Queensland
BL: Burrill Lake, New South Wales
CAN: Canning River, Western Australia
CRBB: Clyde River, New South Wales
CS: Cockburn Sound, Western Australia
CR: Crookhaven River, New South Wales
GI: Gordon Inlet, Western Australia
GL: Gippsland Lakes, Victoria
HI: Hammersley Inlet, Western Australia
HAI: Hardy Inlet, Western Australia
II: Irwin Inlet, Western Australia
LA: Lake Alexandrina, South Australia
LI: Lake Illawarra, New South Wales
MARO: Maroochy River, Queensland
MARY: Mary River, Queensland
MRI: Moore River Inlet, Western Australia
MB: Moreton Bay, Queensland
MOR: Moruya River, New South Wales
MYL: Myall Lakes, New South Wales
OI: Oldfield Inlet, Western Australia
PI: Parry Inlet, Western Australia
PPB: Port Phillip Bay, Victoria
SCR: Scott River, Western Australia
SR: Shoalhaven River, New South Wales
SGB: St George's Basin, New South Wales
STM: St Mary's Inlet, Western Australia
SWR: Swan River, Western Australia
TR: Tomaga River, New South Wales
TOR: Torbay, Western Australia
TL: Tuggerah Lakes, New South Wales
WL: Wallis Lake, New South Wales
WN: Walpole Nornalup, Western Australia
Warnbro Sound, Western Australia
Wellstead Inlet, Western Australia
Wilson Inlet, Western Australia
(Adapted from Heap *et al.,* 2001)

The sediments in Coombabah Lake system exhibit greater TP concentrations than do Oldfield Inlet, Gippsland Lakes, Wallace Lake and Moore River Inlet (Figure 2-7) and the almost pristine Bellinger Estuary system of northern New South Wales (176.6, µg g^{-1} dry wt) (Smith 1996). However, Coombabah sediments have smaller mean concentrations of TP concentrations than those found in urbanised Australian coastal systems such as those of Maroochy River, Beaufort Inlet, Canning River, Mary River, Irwin Inlet and Walpole Nornalup (Figure 2-7).

Coombabah sediments display TP concentrations similar to TP concentrations observed in Leschenault Estuary (Western Australia), 330.7 µg g^{-1} dry wt (Hill *et al.* 1992); Harvey Estuary (Western Australia), 431.0 µg g^{-1} dry wt (Birch *et al*. 1999); Peel Estuary (Western Australia), 206.6 µg g^{-1} dry wt (Birch *et al*. 1999); Richmond River (New South Wales), 431.3 µg g^{-1} dry wt (Smith 1996). These systems are modified systems with nutrient inputs from agriculture and urban runoff non-point source (NPS) inputs.

Coombabah Lake sediments are less impacted from nutrient loading than more heavily urbanised regions within Australia, illustrated by the lower mean TP concentrations. The Brisbane (Queensland), Hawkesbury (central New South Wales) and Swan–Canning (Western Australia) estuaries are heavily urbanised regions located in major metropolitan areas that receive large nutrient loads, characterised by reported TP concentrations of 642.5, 929.0 and 877.0 µg g^{-1} dry wt respectively (Hill *et al*., 1992; Smith, 1996; Birch *et al*., 1999). These systems are not representative of the Coombabah Lake system but rather of heavily urbanised coastal systems elsewhere in Australia.

Lower surface samples collected from grids 13, 15–18, 21, 30 and 56 far exceeded the average TP concentration of 120–450 µg g^{-1} indicating localised pools of potentially available P in the sediment column. The occurrence of elevated phosphorous concentrations within lake sediments is believed to be a result of NPS urban runoff from the catchment and immediate lake area containing P sorbed to colloidal material (clays and hydrous Fe and Al oxides) and soluble organic matter. A golf course development and an adjoining small tributary which encroaches on the lake shoreline at sample grid 10 appear to play an important role in the elevated concentrations of TP within the sediments through the runoff of regular applications of fertilisers and pesticides used in course maintenance (Winter & Dillon 2005). Runoff from urban areas and golf courses is often presumed to be a significant contributor to NPS water pollution originating from the urban environment (Kohler *et al*. 2004).

Observed E-P concentrations within Coombabah Lake were generally low. E-P concentrations showed variability patterns similar to TP at both sample depths, with increased concentrations located within the southern regions of the lake, except for sample grids 52 and 56, occurring in the western arm of the lake. The E-P concentrations relate to the P form of dissolved inorganic phosphorous within the lake sediments and is generally considered to be chemically indicative of orthophosphate (PO_4).

Ammonium was detected at each of the sample grids within the lake. With the exception of sample grids 1–6, 52 and 56, E-NH_4^+ concentrations throughout Coombabah Lake upper and lower surface sediments were very low. The more elevated concentrations of E-NH_4^+ were associated with fine-grained sediments within the lake and sediments at the freshwater entrance of Coombabah Creek. Greater E-NH_4^+ concentrations may result from NPS urban runoff from both the catchment and lake surroundings containing fertilisers, decaying organic matter and faeces from domestic animals and animals inhabiting the lake area.

The present baseline data on nutrient concentrations in surface sediments within Coombabah Lake indicates that the local environment is typical of Australian estuarine and coastal water sediments and suggests that the lake sediments are currently not heavily impacted by urban expansion. To determine what role they play as a source of nutrients requires further information, for example, on the flux of nutrients across the sediment–water interface and on the mechanisms that control uptake and release by bottom sediments of nutrients (including the role of resuspension, settling and decomposition of organic material). Although the obtained data suggests typical coastal and estuarine Australian nutrient concentrations are found within the lake sediments, in view of the observed effects of urban-based influences entering the lake and the increasing urban development within the lake catchment and greater surrounding Moreton Bay region, regular future measurements of sediments should be undertaken to monitor change in nutrient loadings within Coombabah Lake.

Metals and sulfides studies

Sediment quality assessment for metals and sulfides in Coombabah Lake was undertaken in three steps:

1. A scoping study of total metal concentrations and pyrite content across the lake, which were compared against the ANZECC/ARMCANZ (2000) trigger values and Queensland Government guidelines (Dear *et al.* 2002) respectively
2. A risk assessment of monosulfides in sediments in the vicinity of the STP Creek regarding pH, redox conditions, AVS and immediate sediment oxygen demand (ISOD)
3. An integrated iron–sulphur–trace metal study of sediments along a transect from Coombabah Lake upstream into Coombabah Creek, where the scoping study identified higher metal and pyrite levels. Here sediment quality was assessed as follows:
    i. Total metal and pyrite contents were compared against the ANZECC/ARMCANZ trigger values and Queensland Government guidelines (repeating step 1). Total metals were also compared to aluminium levels to assess the degree of trace metal enrichment.
    ii. Total metal concentrations were compared with 1M HCl metal concentrations.
    iii. The sum of molar concentrations of extractable Cd, Cu, Ni, Pb and Zn minus the AVS concentrations was calculated to indicate if AVS is sufficient to sorb the metals.
    iv. The "%AVS/Fe" parameter was calculated to determine if the sediments are highly dynamic with regard to AVS formation and oxidation (and conversion to pyrite).

Scoping study

Metals

Analysis of surface sediment samples from 10 locations across the lake found levels for heavy metals were generally below the ANZECC/ARMCANZ (2000) guidelines (Tables 2-11 and 2-12). Exceptions occurred at Site 1, just downstream of the Coombabah Creek bridge and at Site 8, at the upstream mouth of Coombabah Creek where it meets Coombabah Lake. At Site 1 the nickel level was 25 mg kg^{-1}, slightly above the ANZECC trigger value of 21 mg kg^{-1} and at Site 8, the arsenic level was 20 mg/kg, right at the ANZECC trigger level. Interestingly, samples from Sites 19 and 34, in the vicinity of the sewage treatment plant (STP) and possibly acidic drains, did not show elevated metal concentrations. Two possible explanations are: firstly, that STP and catchment management has effectively retained any metals on site; or, secondly, that metals accumulations may occur below the surface depths sampled. Given the apparently low lake sedimentation rates however (see earlier shell dating section and Chapter 4), this seems unlikely. Based on the scoping study, heavy metal levels appear to be generally low.

The total metal results from the scoping study (Tables 2-11and 2-12) are similar to the low levels (for lead and zinc only) found in five 0–20 cm samples of Coombabah Lake sediments (Dixon & Draper 1996). These authors also found that levels of mercury in sediments were at non-detectible concentrations.

Some of the variation in the scoping study results can be explained by differences in particle size, with the sandier samples from the northern tidal delta of the lake (e.g. Sites 59, 44 and 31) having lower metal levels. Interestingly, low metal levels were also found in sediments located in waters draining a catchment that includes the STP and the Suntown landfill.

Total metal levels in sediments do not represent the “potentially bioavailable” metal content and are likely to overestimate the risk. Given the generally low total metal levels found, it is unlikely that metal levels in Coombabah Lake sediments are of immediate concern in terms of bioaccumulation, although the results are a warning sign of a possible increase in some metals in sediments derived from the upper Coombabah Creek catchment. This small catchment is urbanised, with potential sources of metal contaminants including an industrial estate, residential development, a major highway and pipeline and a golf course.

This issue was investigated further in the integrated sulfur, iron and trace metal case study, and is discussed later in this section.

Table 2-11. Total metal analysis of lake sediments: manganese, molybdenum, sodium, nickel, phosphorus, lead, zinc, sulfur and chromium-reducible sulfur

Site	Depth cm	Manganese mg kg^{-1}	Molybdenum mg kg^{-1}	Sodium %	Nickel mg kg^{-1}	Phosphorus %	Lead mg kg^{-1}	Zinc mg kg^{-1}	Sulfur %	Sulfur; chromium reducible %
1	0–2	180	<3	1.20	**25****	0.033	7.4	58	1.20	0.817
8	0–2	120	<3	0.96	14	0.040	<5	46	0.64	0.431
19	0–2	160	<3	1.50	17	0.051	7.2	57	0.56	0.344
20	0–2	81	<3	1.00	9.3	0.025	<5	36	0.41	0.245
22	0–2	140	<3	0.96	13	0.032	5	43	0.34	0.196
31	0–2	85	<3	0.68	8.8	0.018	<5	29	0.21	0.123
34	0–2	140	<3	1.40	15	0.034	<5	49	0.57	0.362
44	0–2	130	<3	0.94	9.7	0.023	<5	39	0.34	0.203
56	0–2	120	<3	1.50	15	0.037	5.3	47	0.51	0.294
59	0–2	78	<3	0.42	4	<0.010	<5	17	0.15	0.087
ANZECC Trigger					**21**		**50**	**200**		**0.1**

**** At or above ANZECC/ARMCANZ (2000) guidelines**

Table 2-12. Total metal analysis of lake sediments: aluminium, arsenic, calcium, cobalt, chromium, copper, iron, potassium and magnesium

Site	Depth cm	Aluminium %	Arsenic mg kg^{-1}	Calcium %	Cobalt mg kg^{-1}	Chromium mg kg^{-1}	Copper mg kg^{-1}	Iron %	Potassium %	Magnesium %
1	0–2	3.4	13	0.25	11.0	32.0	6.4	3.40	0.49	0.71
8	0–2	2.5	**20****	0.60	10.0	25.0	2.3	2.50	0.37	0.52
19	0–2	2.9	13	0.42	9.9	31.0	8.0	3.00	0.41	0.69
20	0–2	1.5	10	0.14	7.0	15.0	1.4	1.60	0.23	0.38
22	0–2	2.2	7.6	0.29	8.4	24.0	3.6	2.10	0.31	0.48
31	0–2	1.2	5.3	0.57	5.8	15.0	4.2	1.30	0.19	0.32
34	0–2	2.6	7.7	0.41	8.4	27.0	6.6	2.50	0.38	0.59
44	0—	1.8	7.3	0.20	7.6	20.0	3.7	1.80	0.27	0.43
56	0—	2.6	9.2	0.38	8.6	29.0	5.3	2.40	0.39	0.60
59	0-2	0.65	5.1	0.08	3.9	7.8	<1.2	0.75	0.10	0.18
ANZECC Trigger			**20**			**80**	**65**			

****At or above ANZECC/ARMCANZ (2000) guidelines**

Pyrite

Results of the scoping study are presented in Table 2-11. The chromium reducible sulfur tests show that about 60–70% of total S in the samples was made up of sulfides, believed to be mostly in the form of pyrite. This is the mineral predominantly responsible for the problems that characterise acid sulfate soils (Dent 1986). Except for Site 59, the levels of chromium reducible sulfur were all above the action level of 0.1% where, in Queensland, management action is required should they be disturbed for development (Dear *et al*. 2002). Levels were highest in the fluvial delta sediments at the entrance of upper Coombabah Creek and lowest in the sandier tidal delta sediments where the lake exits.

The levels of chromium reducible sulfur recorded (Table 2-11) are sufficient to trigger further acid sulfate soil (ASS) investigation and management action should the sediments be disturbed. However they are similar to the values obtained for partly oxidised ASS commonly found on the Gold Coast coastal plains (Manders *et al*. 2002). They are much lower than the values (>1%) commonly found in unoxidised (potential) ASS on the Gold Coast.

The remaining sulfur in the sediments is likely to be in organic form. The presence of sulfides in the surface sediments could have consequences for water quality of the lake. If AVS comprise a significant component of sulfide S, there is potential for their resuspension by wave action to lead to rapid deoxygenation of the lake waters.

The risks of AVS to lake water quality are further discussed in the following section.

Monosulfide geochemistry study—STP Creek

Resuspension of sediments containing significant concentrations of iron monosulfides poses a risk to water quality (Sullivan & Bush 2000). Oxidation of any suspended iron monosulfides consumes dissolved oxygen, produces Fe(II) and zero valent S. Acidification occurs after the dissolved oxygen concentration rises, allowing oxidation of Fe(II) to Fe(III) and S to SO_4.

Thick deposits of iron monosulfides in drains and protected upper reaches of estuary tributaries in ASS areas of northern New South Wales have led to rapid deoxygenation and acidification of waterways (Bush *et al*. 2004). Mapping by (Manders *et al*. 2002) found that the wetlands surrounding Coombabah Lake contain ASS layers within a metre of the surface and Paice (1996) observed that acidic pHs have been periodically recorded near the STP Creek. Sediment samples were taken from the STP Creek area to test if an iron monosulfide risk of lake deoxygenation and acidification was present. Testing took the form of measurements of pH, Eh, AVS and immediate (5 min) ISOD.

The results in Tables 2-13 and 2-14 reflect typical suboxic-to-anoxic sediment, which is generally not sufficiently reducing for widespread sulfate reduction to sulfides. Hence the level of AVS present in these sediments is notably low compared to other values recorded in New South Wales (e.g. Bush *et al*. 2004) and Queensland (e.g. Burton *et al*. in press, a) estuaries. They are also low compared with the value of about 0.3% typically found for estuarine sediments by Morse and Cornwell (1987). However, sulfate reduction to AVS will occur in microniches. For example, the highest AVS level of 0.023% was observed in the 2–4 cm depth in the STP creek channel (a small volume of black sediment observed in the field). The AVS in this depth was associated with a small piece of rotting bark/timber (which wasn't used in the AVS analyses).

A stability field diagram for the Fe–S system (taken from Langmuir 1997) is presented in Figure 2-8. The stability field for pyrite is not shown, but would cover the FeS field and some of the Fe^{2+} field. The plots show that the pH–Eh conditions prevailing are well outside the stability fields for FeS. Thus, any FeS forming in microniches will tend to dissolve (by oxidative dissolution) with the resultant Fe^{2+} diffusing away from the FeS particle. Importantly, the stability fields refer to equilibrium conditions and say nothing about the rates of AVS (or pyrite) oxidation.

Compared to other measurements in the estuaries and drains of New South Wales, the ISOD values are low and reflect the low level of AVS measured in the sediment samples, indicating that there is no significant risk to the level of dissolved oxygen in lake waters from the resuspension of sediments.

Table 2-13. Selected properties of STP Creek sediments to assess monosulfide risk

Site 1 (in channel at mouth of STP Creek):

Depth (cm)	pH	Eh	Water content	Mean AVS* (%)	SD AVS (%)	ISOD** (5 min) (mg O_2 g^{-1})
0–2	6.90	50	0.71	0.0014	0.0005	0.054
2–4	7.09	-24	0.42	0.0231	0.0209	0.034
4–6	7.39	20	0.43	0.0091	0.0016	0.040
6–8	7.39	-6	0.38	0.0026	0.0002	0.044
8–10	7.42	16	0.35	0.0007	0.0010	0.038
10–14	7.32	-16	0.34	0.0000	0.0000	
14–18	7.38	49	0.34	0.0002	0.0003	
18–22	7.40	-70	0.32	0.0000	0.0000	

Site 2 (exposed sediment at mouth of STP Creek):

Depth (cm)	pH	Eh	Water content	Mean AVS (%)	SD AVS (%)	ISOD (5 min) (mg O_2 g^{-1})
0–2	7.34	18	0.51	0	0	0.056
2–4	7.30	–6	0.44	0.0026	0.0002	0.016
4–6	7.29	–30	0.40	0.0105	0.0115	0.044
6–8	7.31	–45	0.37	0.0031	0.0009	0.022
8–10	7.38	–41	0.34	0.0017	0.0025	0.042
10–14	7.40	–38	0.32	0.0000	0.0000	
14–18	7.42	–40	0.31	0.0013	0.0018	
18–22	7.40	–53	0.34	0.0000	0.0000	

Site 3 (deeper channel):

Depth (cm)	pH	Eh	Water content	Mean AVS (%)	SD AVS (%)	ISOD (5 min) (mg O_2 g^{-1})
0–2	7.66	60	0.26	0.0014	0.0005	0.018
2–4	7.54	30	0.24	0.0002	0.0000	0.022
4–6	7.58	40	0.22	0	0	0.018
6–8	7.71	17	0.26	0	0	0.020
8–10	7.68	13	0.27	0	0	0.026
10–14	7.59	26	0.29	0	0	
14–18	7.61	-14	0.29	0	0	
18–22	7.50	-37	0.31	0	0	

* AVS = acid volatile sulphide
** ISOD = immediate (5 min) sediment oxygen demand

Table 2-14. Selected properties relating to iron and sulphur geochemistry of lake sediments

Site 4:

Depth (cm)	pH	Eh	Porewater Fe(II) (mg l^{-1})	Porewater S(II) (mg l^{-1})	Porewater SO4 (mg l^{-1})	AVS (µmol g^{-1})	Elemental S (µmol g^{-1})	Pyrite-S (µmol g^{-1})	Total S (%)	Total C (%)
0–3	6.79	110	10.98	0.00	2210	0.07	0.43	61.0	0.18	1.01
3–6	6.85	77	9.83	0.00	2040	2.86	2.30	82.0	0.32	0.97
6–9	7.05	71	3.22	0.00	2000	5.04	5.99	112.4	0.36	1.05
9–12	7.05	79	4.69	0.04	2110	2.07	1.95	151.0	0.54	1.24
12–15	7.10	113	2.20	0.03	2090	1.44	1.79	183.9	0.56	1.32
15–20	7.26	121	0.34	0.01	2250	0.30	0.91	278.7	0.90	1.54
20–25	7.22	123	0.06	0.05	2220	0.06	0.88	316.8	0.94	1.84
25–30	7.19	138	0.02	0.11	2230	0.19	2.48	369.2	1.19	1.27

Site 5:

Depth (cm)	pH	Eh	Porewater Fe(II) (mg l^{-1})	Porewater S(II) (mg l^{-1})	Porewater SO4 (mg l^{-1})	AVS (µmol g^{-1})	Elemental S (µmol g^{-1})	Pyrite-S (µmol g^{-1})	Total S (%)	Total C (%)
0–3	6.88	66	7.76	0.00	2120	0.09	0.38	95.5	0.30	1.70
3–6	6.81	64	20.35	0.00	2070	1.72	2.99	156.2	0.52	2.36
6–9	6.92	75	10.30	0.00	2090	2.98	4.61	160.0	0.50	1.81
9–12	6.90	69	9.95	0.04	2020	2.75	3.91	204.6	0.62	1.72
12–15	6.84	71	4.83	0.04	2100	0.93	2.28	333.3	1.00	2.28
15–20	7.18	97	0.44	0.05	2040	0.72	2.16	421.1	1.21	2.54
20–25	7.26	120	0.05	0.03	1870	0.97	2.29	364.3	1.12	2.46
25–30	7.42	53	0.07	0.29	1480	0.73	1.24	340.1	1.21	1.90

Site 6:

Depth (cm)	pH	Eh	Porewater Fe(II) (mg l^{-1})	Porewater S(II) (mg l^{-1})	Porewater SO4 (mg l^{-1})	AVS (µmol g^{-1})	Elemental S (µmol g^{-1})	Pyrite-S (µmol g^{-1})	Total S (%)	Total C (%)
0–3	6.44	90	6.12	0.00	1920	0.00	1.19	210.2	0.65	2.70
3–6	6.76	30	14.72	0.00	1950	0.88	2.01	238.8	0.69	2.44
6–9	6.98	47	6.78	0.00	2020	2.16	2.27	305.5	0.88	2.36
9–12	7.06	73	2.61	0.00	1720	1.85	3.95	475.6	1.46	2.33
12–15	7.47	78	0.46	0.00	1840	0.65	0.73	401.7	1.10	2.28
15–20	7.57	95	0.05	0.00	2010	0.36	2.18	371.9	1.19	1.97
20–25	7.54	101	0.00	0.02	1980	0.28	2.12	386.0	1.16	2.36
25–30	7.54	100	0.01	0.00	1870	0.25	1.63	380.5	1.19	2.28

Site 9:

Depth (cm)	pH	Eh	Porewater Fe(II) (mg l^{-1})	Porewater S(II) (mg l^{-1})	Porewater SO4 (mg l^{-1})	AVS (µmol g^{-1}	Elemental S (µmol g^{-1})	Pyrite-S (µmol g^{-1})	Total S (%)	Total C (%)
0–3	6.68	49	59.92	0.00	1800	0.00	0.82	143.3	0.32	2.71
3–6	6.64	49	76.80	0.00	1880	12.64	8.74	140.1	0.39	2.73
6–9	6.70	72	34.77	0.00	1920	21.58	10.51	180.3	0.41	2.49
9–12	6.67	71	39.64	0.00	2060	24.81	16.17	161.3	0.47	2.38
12–15	6.52	72	37.97	0.00	1050	14.99	10.20	180.6	0.45	2.33
15–20	6.70	68	21.12	0.00	1120	19.57	15.59	175.2	0.47	2.32
20–25	7.05	66	8.37	0.00	1140	8.02	9.06	204.3	0.46	2.22
25–30	7.10	52	1.27	0.00	1070	4.30	5.64	193.5	0.55	2.33

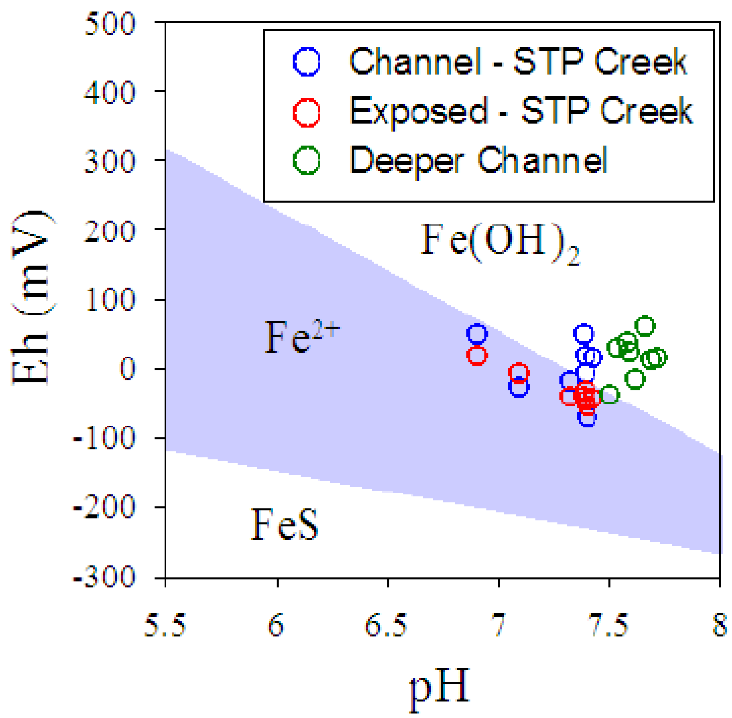

Figure 2-8. Stability field diagram (source: Langmuir 1997) for the Fe–S system of STP Creek sediments, Coombabah Lake

Integrated sulfur, iron and trace metal study

Six sample sites (Sites 4 to 9, Figure 2-1) along a transect were sampled from the lake upstream to upper Coombabah Creek. Sediment columns from Sites 4, 5, 6 and 9 were subjected to detailed laboratory testing and selected results are presented in Table 2-14. Sediment pH for all four cores was near-neutral, spanning pH 6.44–7.57 at Site 6. The measured Eh values were moderately reduced below 15 cm depth at Site 4 within the lake whereas the other cores, confined in the quieter reaches of Coombabah Creek, were more reduced but not strongly so (generally <30–100 mV). No cores contained living mangrove roots, which promote oxic conditions within mangrove sediment profiles (Clark *et al*. 1998).

Total C contents ranged from 1 to 2.7% in near-surface sediment, maintaining levels within this range to a depth of 30 cm below the sediment–water interface (Table 2-14). Compared to the wave-exposed lake sites (Site 4 and the results of the overview survey), levels of total C (0.1–1.8%) were higher at Sites 5, 6 and 9 (1.7–2.7%), located within the protected confines of Coombabah Creek.

Sulfur and iron fractionation

Pore-water S(II) concentrations were negligible or very low (up to 0.11 mg l^{-1}). Pore-water Fe(II) concentrations were moderately high near the sediment–water interface (up to 76.8 mg l^{-1} at Site 9), decreasing to lower levels with depth. This trend may be attributed to (i) production of Fe(II) by reductive dissolution of ferric minerals in near-surface sediment and (ii) progressive sequestration of this Fe(II) due to pyrite formation at greater depths.

Low maximum AVS concentrations of 5.0 µmol g^{-1}, 3.0 µmol g^{-1} and 2.2 µmol g^{-1} were present in near-surface sediment (6–9 cm) at Sites 4, 5 and 6 respectively. Site 9 (upstream creek reach) was quite distinct in having moderately high AVS concentrations in all sections from 3 cm to 30 cm, peaking at 24.8 µmol g^{-1} in 9–12 cm deep sediment. These concentrations are comparable to those presented in Burton *et al*. (2005b), in surface sediments from sites adjacent to other urbanised areas of Moreton Bay. There was a decrease in AVS concentrations with depth below 6–9cm, with <1µmol g^{-1} observed below 15 cm, except for Site 9. Site 9 decreased to 4.3 µmol g^{-1} and has similar values to Site (W2), a small drainage depression surrounded by mangroves, 20 m from intertidal mudflats in Moreton Bay (Burton *et al.* in press, a).

Elemental S was most abundant (12.6–24.8 µmol g^{-1}) at Site 9 in the layers sampled between 3 cm and 20 cm. Elemental S levels at Sites 4, 5 and 6 are much lower (<6 µmol g^{-1}) and peak around 6–12 cm. These concentrations and profile trends are comparable to the corresponding AVS data. This indicates *in situ* oxidation of AVS species such as pore-water sulfide or iron monosulfides (Troelsen & Jørgensen 1982). Such oxidation processes are known to produce intermediate S species (i.e. aqueous polysulfides) that are necessary for efficient pyrite formation in suboxic sediments. This probably explains the generally low levels of AVS and coexisting abundance of pyrite-S in the sediment profiles described here. As expected, the pyrite content at all sites generally increased with depth below the sediment–water interface.

Considerably lower concentrations of pyrite were present at the 0–3 cm layer at Sites 4 and 5, with pyrite-S levels of 61 and 96 µmol g^{-1} respectively. These trends in AVS to pyrite-S ratios may be caused by formation of pyrite from meta-stable iron-monosulfide precursors (Schoonen & Barnes 1991). This process would lead to depletion of AVS in deeper, older sediments and a coexisting relative enrichment of pyrite-S. The higher AVS to pyrite-S ratios at Site 9 suggest that these sediments are younger and the AVS has not yet been depleted. Alternatively, the moderately high levels of pore-water Fe(II) at this site may retard rates of pyrite formation, by rapidly sequestering pore-water sulfide by iron-monosulfide precipitation (Burton *et al*. in press, b).

Risk assessment

Total metals

Total metal concentrations are presented in Figures 2-9 and 2-10. All total metal values are well below the ANZECC/ARMCANZ sediment quality sediment trigger levels except for arsenic and nickel, which are close to the respective trigger values in some depth intervals for Sites 6 and 9. This reinforces the findings of the scoping study which found in upper Coombabah Creek that one surface sediment sample exceeded nickel trigger values (Site 1 in Table 2-13) and another sample exceeded the arsenic trigger value (Site 8 in Table 2-13).

It is possible that the total concentrations of metals could be explained by a higher abundance of aluminosilicate minerals in the fine fraction. Aluminium is a useful reference element as it is present as a stable element and human development activities add insignificant amounts compared to its natural abundance in sediments (Burton *et al*. 2005a). Burton *et al*. (2005a) found fine sediments (<63 µm) to be strongly correlated with "near total" aluminium. This "normalisation approach" was used by Burton *et al*. (2005a) to assess the degree of trace metal enrichment of sediments in the Southport Broadwater.

The results for Coombabah Lake in Figure 2-11 suggest that the concentrations of most of the metals tested may be contained within these fine fractions. The degree to which variation in metal concentrations is explained by the concentration of Al is shown by the following correlation coefficient (r) values: As 0.65, Cd 0.48, Cr 0.96, Cu 0.89, Fe 0.94, Fe 0.60, Mn 0.62, Ni 0.96, Pb 0.91 and Zn 0.97. Aluminium and trace metal levels also show a consistent increasing trend from Site 4 within the lake, upstream to Coombabah Creek (Site 9). The trends of increasing metal concentrations can be largely explained by natural gradients in the abundance of aluminosilicate minerals. Burton *et al*. (2005a) came to similar conclusions for the Southport Broadwater except for enclosed canal systems with poor flushing conditions, where fine organic-rich sediments from urban stormwater accumulate or at marinas with dry-dock facilities. Such conditions do not occur in Coombabah Lake or its creek tributary.

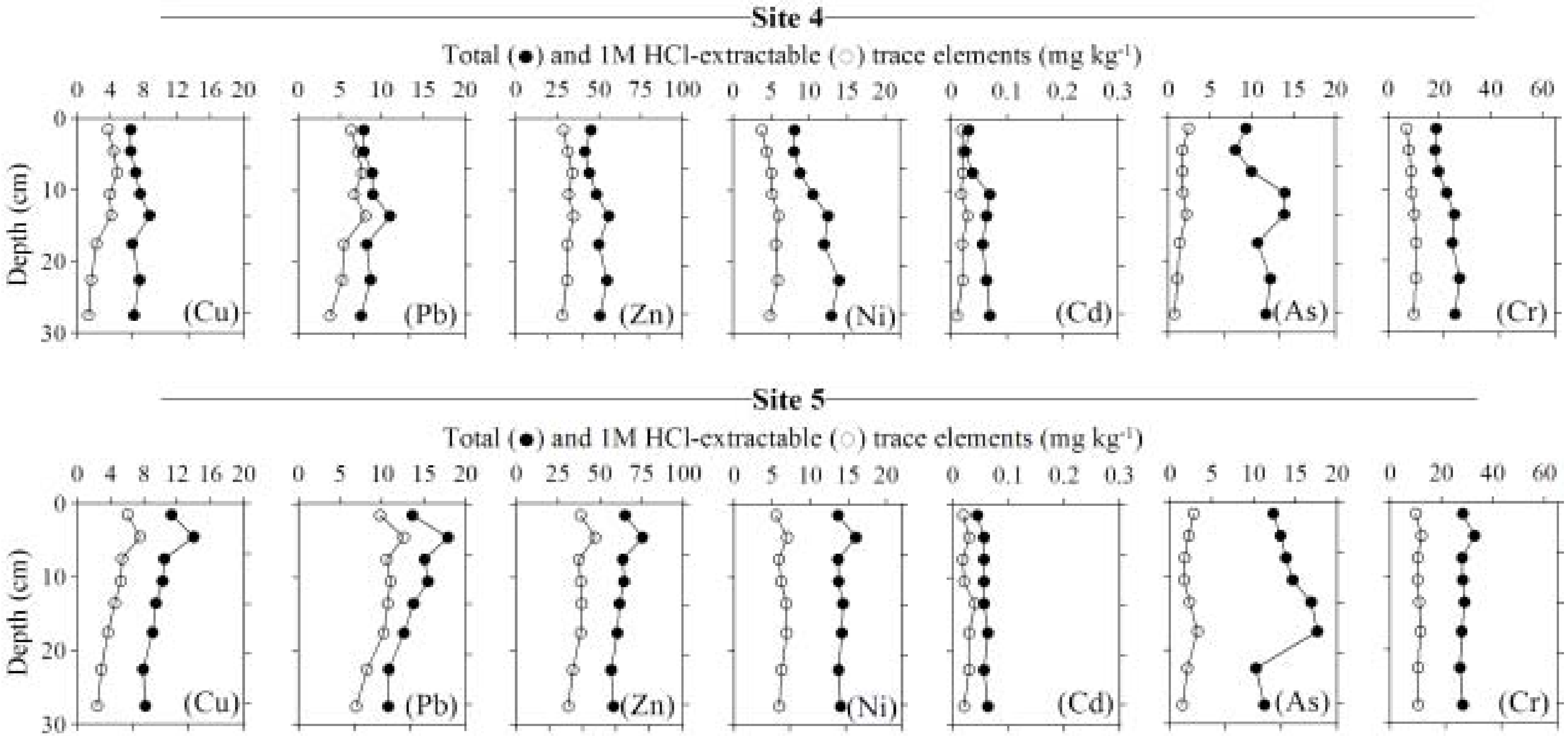

Figure 2-9. Sediment profiles of total and 1M HCl-extractable trace element concentrations at Sites 4 and 5 in Coombabah Lake

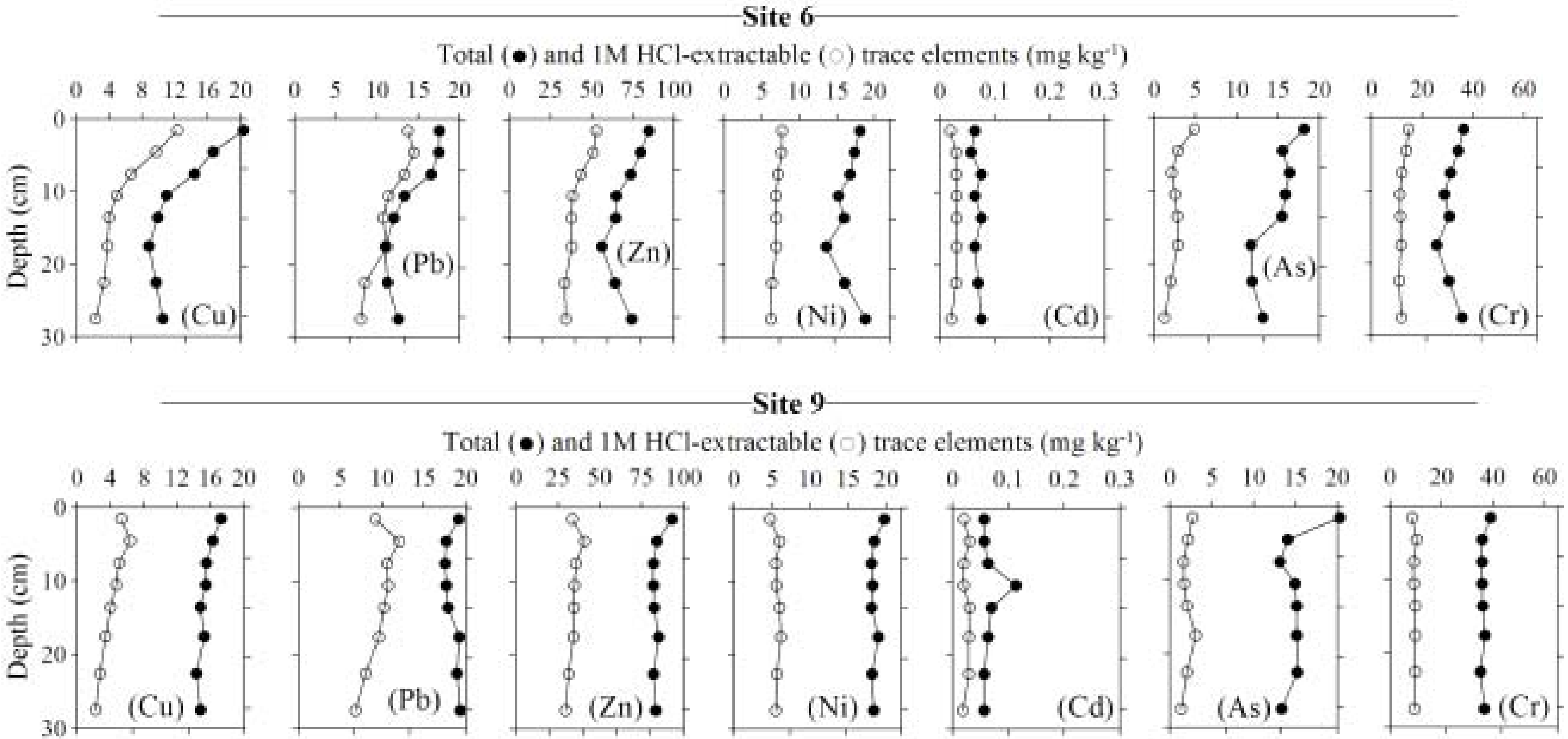

Figure 2-10. Profiles of total and 1M HCl-extractable trace element concentrations at Sites 6 and 9 in Coombabah Lake

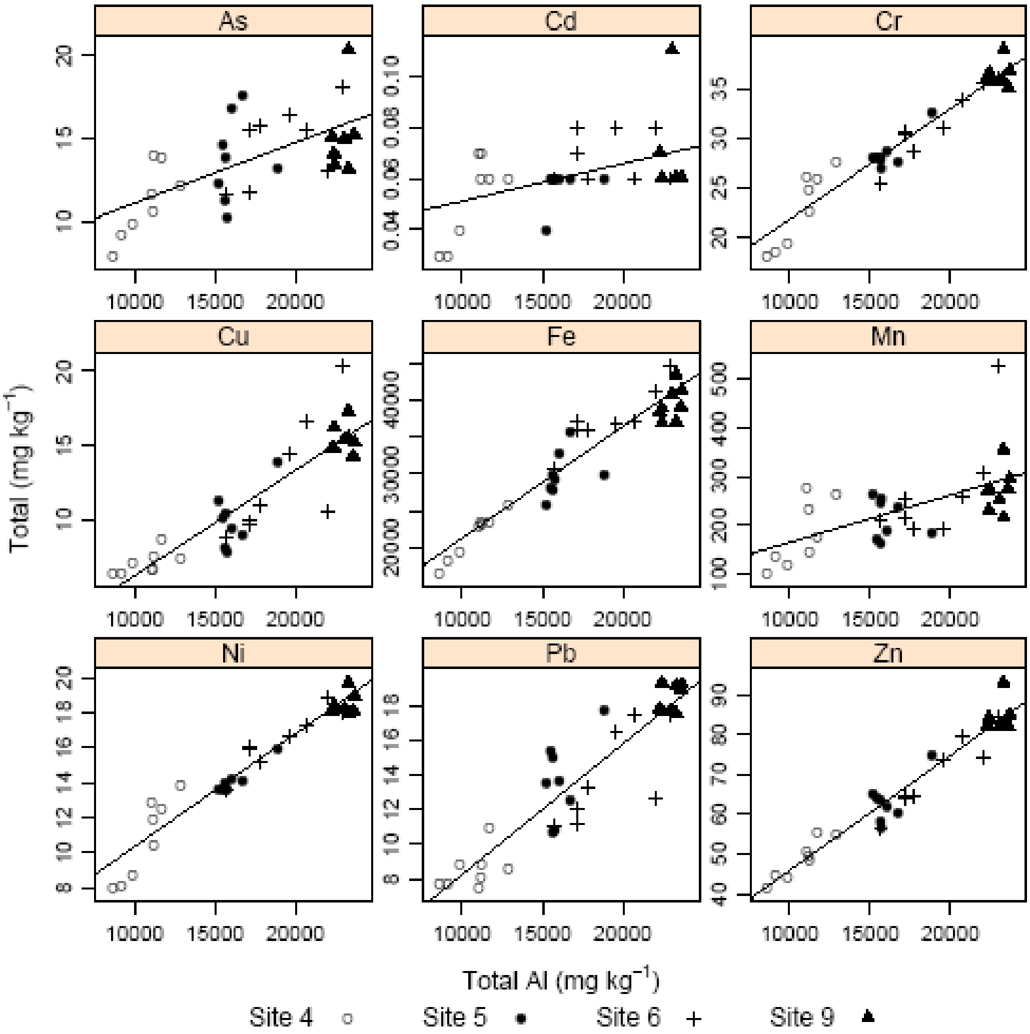

Figure 2-11. Relationship between aluminium (used as a proxy for fine alumino-silicate minerals) and other metals at Sites 4, 5, 6 and 9 in Coombabah Lake

Bioavailable metals

Total metal concentrations are compared with 1M HCl metal concentrations down the four sediment columns in Figures 2-9 and 2-10. The extracts of HCl attempt to measure metal availability and provide a more reliable assessment of sediment quality (ANZECC/ARMCANZ, 2000).

For Cd, HCl-extract concentrations were comparable with total concentrations, suggesting total values may be an adequate indicator for this metal. Pb concentrations are also comparable except at Site 9, where total concentrations are consistently highest of all the sites but the HCl-extract fraction is a smaller component. With all the other metals the HCl-extract concentrations were low compared to the corresponding total metal concentrations, but, except for As, all seem to vary in proportion to the total value. The relationship between

HCl-extractable As and total As was the most variable of all the metals examined. This is probably due to processes such as As sorbtion/co-precipitation with pyrite or by precipitation of arsenopyrite as suggested by Burton *et al*. (in press, a).

The 1M HCl-extractable concentrations of As, Cr, Cu, Ni, Pb and Zn were all strongly correlated with 1M HCl-extractable Fe (Table 2-15). The 1M HCl extraction procedure provides a measure of the abundance of reactive Fe minerals, such as poorly crystalline Fe(III)-oxyhydroxides and Fe(II)-monosulfides. Given the positive Eh values and very low AVS levels reported here, these reactive Fe minerals can be assumed to mostly comprise Fe(III)-oxyhydroxides. Fe(III)-oxyhydroxide minerals are known to be a major sorbent for As, Cr, Cu, Ni, Pb and Zn in suboxic sediments with low AVS concentrations (Chapman *et al*. 1998). This interaction may largely explain the observed correlation between 1M HCl-Fe and the 1M HCl-extractable heavy metals.

Table 2-15. Pearson correlation coefficients for relationships between 1M HCl-extractable trace metals and selected sediment properties

	As	Cd	Cr	Cu	Ni	Pb	Zn
Total concentration	**0.524****	0.0944	0.2951	**0.635****	0.434*	**0.646****	0.397*
AVS	-0.0893	0.0345	-0.3290	-0.0080	-0.1557	0.1817	-0.0785
Pyrite-S	-0.0190	0.3014	**0.522****	-0.2934	**0.546****	0.0036	0.0230
AVS/pyrite-S	-0.1046	-0.0035	-0.367*	0.0112	-0.2063	0.1620	-0.0820
Total C	0.409*	**0.465****	**0.490****	0.3381	**0.619****	**0.629****	**0.495****
1M HCl-extractable Fe	**0.671****	0.2516	**0.713****	**0.844****	**0.693****	**0.809****	**0.864****

n = 32
* P<0.05
** P<0.01
Significant correlations with P <0.01 are shown in bold

Sulfide-bound metals

The sulfide content of sediments is a key modifier of bioavailability of some metals (Simpson *et al.,* 2005). Trace metals such as Cd, Cu, Ni, Pb and Zn form sulfide minerals that are stable under reducing conditions and have very low solubility (Chapman *et al*.1998). The sum of molar concentrations of extractable Cd, Cu, Ni, Pb and Zn minus the AVS concentrations presented in Table 2-15 indicates that AVS is sufficient to sorb the metals. Positive values indicate insufficient AVS is present to sorb the corresponding summed metal concentrations. However, this does not appear to represent a risk in Coombabah

Lake as the extractable metal concentrations are all low. According to the CSIRO *Handbook of sediment quality assessment* (Simpson *et al*. 2005), an SEM-AVS value of 5 µmol g^{-1} is a useful trigger value.

Sulfide stability

The "%AVS/Fe" values in Table 2-16 are all considerably less than 20%, which indicates that the sediments are highly dynamic with regard to AVS formation and oxidation (and conversion to pyrite). This is consistent with the generally low AVS concentrations and the corresponding concentrations of elemental sulfur (which forms as an AVS oxidation product and is necessary for pyrite formation in suboxic sediments). This suggests that AVS may represent a relatively transient binding phase for trace metals in Coombabah Lake.

Every assessment confirmed that overall the bioavailability risk from metals within Coombabah Lake sediments is currently low. However the sediments have low and generally transient concentrations of AVS, which means that the lake sediments may not be able to tolerate significant additional metal sources and buffer the system sufficiently to prevent metals bioaccumulating following an influx.

Table 2-16. Metal risk assessment properties for Coombabah Lake sediments

Site 4:

Depth (cm)	SEM (μmol g^{-1})	AVS (μmol g^{-1})	SEM-AVS	%AVS/Fe
0–3	0.59	0.07	0.52	0.05
3–6	0.66	2.86	–2.20	1.92
6–9	0.72	5.04	–4.32	3.11
9–12	0.67	2.07	–1.40	1.37
12–15	0.74	1.44	–0.71	0.92
15–20	0.63	0.30	0.33	0.21
20–25	0.62	0.06	0.56	0.04
25–30	0.55	0.19	0.36	0.13

Site 5:

Depth (cm)	SEM (μmol g^{-1})	AVS (μmol g^{-1})	SEM-AVS	%AVS/Fe
0–3	0.82	0.09	0.73	0.05
3–6	1.02	1.72	–0.70	0.86
6–9	0.81	2.98	–2.17	1.56
9–12	0.83	2.75	–1.92	1.44
12–15	0.84	0.93	–0.09	0.46
15–20	0.81	0.72	0.10	0.39
20–25	0.71	0.97	–0.26	0.55
25–30	0.65	0.73	–0.08	0.44

Site 6:

Depth (cm)	SEM (μmol g^{-1})	AVS (μmol g^{-1})	SEM-AVS	%AVS/Fe
0–3	1.20	0.00	1.20	0.00
3–6	1.13	0.88	0.25	0.35
6–9	0.95	2.16	–1.21	1.08
9–12	0.83	1.85	–1.02	0.98
12–15	0.80	0.65	0.16	0.35
15–20	0.81	0.36	0.45	0.20
20–25	0.71	0.28	0.43	0.15
25–30	0.70	0.25	0.45	0.13

Site 9:

Depth (cm)	SEM (μmol g^{-1})	AVS (μmol g^{-1})	SEM-AVS	%AVS/Fe
0–3	0.72	0.00	0.72	0.00
3–6	0.88	12.64	–11.76	6.51
6–9	0.77	21.58	–20.81	11.04
9–12	0.75	24.81	–24.06	13.20
12–15	0.74	14.99	–14.25	7.88
15–20	0.73	19.57	–18.85	11.04
20–25	0.65	8.02	–7.37	4.59
25–30	0.61	4.30	-3.69	2.48

Shell dating

Samples were collected from lake sediments (Site 4) and mangrove ponds (Sites EAS 403, EAS 404, EAS 406 and EAS 409, see Table 2-17) that are intermittently tidally exchanged on big tides. Results for all sites are presented in Table 2-17.

The lake and its surrounding mangrove wetlands are known to contain a dense layer of shell consistently found at 0.3–0.5 m below the sediment surface. Such layers would represent a period of hiatus in sediment accumulation when environmental conditions in the lake suited shell populations, and shells were mixed, winnowed and sorted by wave and tidal action. Within the lake (Site 4) the *Spigula trigonella* samples, a deposit feeder, proved to be unsuitable for radiocarbon dating because of its dietary preferences. The *Batillaria australis* results are considered to be valid and suggest a period of at least 300 years over which these conditions may have occurred.

The results may be not so useful for estimating sedimentation rates, but the suggestion can be made that there has been very slow sedimentation in the past 2000 years and that the sediment tends to stay and be reworked within the lake. The heavier shells are the lag deposits retained in the lake by this reworking. Using the lead 210 dates of surface sediments and other evidence, Frank and Fielding (2004) came to similar conclusions. This possibly reflects the estuarine maturity of the system whereby sedimentation rates slow as the lake becomes progressively filled and shallower. There is no evidence in this study to suggest that urbanisation has impacted significantly on sedimentation rates.

Table 2-17. Radiocarbon dates of shells from Coombabah Lake sediments

Site number and location	Sample number	Depth (m)	Shell species	% Modern	δ^{13}C‰	Radio carbon date
EAS403	WK18084	0.5–0.6	*Batillaria australis*	75.4 ± 0.4	-0.1 ± 0.2	2266 +/– 45 BP
27°53'27" E,			*Austrocochlea constricta*			
153°20'35" N			*Corbula c.f. coxi*			
	WK18085	1.3–1.5	*Batillaria australis*	68.5 ± 0.6	-0.3 ± 0.2	3044 +/– 66 BP
			c.f. Anodontia			
			Corbula coxi			
EAS404	WK18086	0.3–0.4	*Batillaria australis*	76.5 ±0.5	-0.7 ± 0.2	2154 +/– 47 BP
27°53'28" E,			*Corbula coxi*			
153°20'36" N	WK18087	0.6–0.8	*Batillaria australis*			
			Corbula c.f. coxi			
			F. Ostreidae sp.	71.1 ± 5.9	-2.1 ± 0.2	2741 +/– 67 BP
EAS406	WK18088	0.8–1.0	Unknown			
27°53'33" E,			*Anadaria c.f. trapezia*	63.6 ± 0.5	-1.0 ± 0.2	3640 +/– 58 BP
153°20'38" N			*Batillaria australis*			
	WK18089	1.4–1.6	*Batillaria australis*			
			Anadaria addita	63.5 ± 0.4	-0.8 ± 0.2	3651 +/– 55 BP
			Corbula c.f. coxi			
EAS409	WK18090	0.5–0.6	*Batillaria australis*	71.4 ± 0.6	-0.2 ± 0.2	2707 +/– 65 BP
27°53'34" E,						
153°20'40" N						
Site 4	WK18091	0.1–0.2	*Spigula trigonella*	68.8 ± 0.8	-1.6 ± 0.2	3009 +/– 95 BP
27°54'40" E,			*Batillaria australis*			
153°20'55" N			Unknown			
	WK18092	0.2–0.3	*Spigula trigonella*	61.9 ± 0.8	-1.9 ± 0.2	3849 +/– 107 BP
	WK18093		*Batillaria australis*	77.9 ± 0.7	-0.8 ± 0.2	2007 +/– 76 BP
			Corbula c.f. coxi			
	WK18094	0.3–0.4	*Batillaria australis*	81.0 ± 1.2	-1.3 ± 0.2	1690 +/–121 BP
			F. Ostreidae sp.			
			Corbula coxi			
			Monilea callifera			
	WK18090	0.4–0.5	*Batillaria australis*			
			Nassarius burchardi			

Conclusion

These investigations revealed high variability within the lake for the measured sediment parameters. A lower variability was observed within the benthic sediment parameters sampled from 2–5 cm below the sediment surface compared to the above-lying 0–2 cm sediment samples. The results also indicate that the physical parameters of the sediment are more stable and less variable across the lake floor than the majority of selected biological and chemical-based sediment parameters.

Most sediment parameter characteristics and variability in the lake are explained by typical estuary processes. Fluvial delta sediments of finer grain size and richer in organic matter derived from freshwater upper catchment sources have accumulated in upper Coombabah Creek and the southern end of the lake. Sandy tidal delta sediments sourced from Moreton Bay are distributed around the northern end of the lake near the mouth of the lower Coombabah Creek.

Pyrite and kaolinite were detected in both sample depths in samples characterised by small grain sizes and occurring within the freshwater entrance region of the lake. At the northern marine end of the lake, sediments comprise mainly quartz and halite. Decreases in sediment grain size coincided with increases in organic matter content.

Higher % C and N values were observed for sediments with greater contributing values of fine sediments within the lake system. Enriched $\delta^{15}N$ values were predominantly found at the entrances of the lake at both the marine and freshwater ends. Measured $\delta^{13}C$ values of surficial sediments were similar and within the range reported for mangroves of -24 to -30 ‰.

On average, TP concentrations observed within the lake compared well with concentrations encountered within Australian estuarine systems. Lower surface samples collected from grids exceeded the average TP concentration indicating localised pools of potentially available P in the sediment column. The occurrence of elevated phosphorous concentrations within lake sediments is believed to be a result of non-point source urban runoff from the catchment and immediate lake area. The P is expected to be sorbed to colloidal material (clays and hydrous Fe and Al oxides) and soluble organic matter.

Increased concentrations for all nutrients analysed were observed in both depths dominated by high mud content, occurring predominantly in the southern region of the lake. Average concentrations in sediments from the southern sample grids (grids 1–25) were greater for all nutrients measured compared to sediments from the northern sandier half of the lake (grids 27–59), where sediments originate from the coastal zone.

The present baseline data on nutrient concentrations in surface sediments within Coombabah Lake indicates that the local environment is typical of Australian estuarine waters and is currently not heavily affected by urban expansion, but future monitoring is required to assess if nutrients are accumulating within the lake system.

To determine to what extent the sediments act as a source of nutrients requires further information, for example, on the flux of nutrients across the sediment–water interface; and on the mechanisms that control uptake and release by bottom sediments of nutrients, including the role of resuspension, settling and decomposition of organic material. Although the obtained data suggests typical coastal and estuarine Australian nutrient concentrations are found within the lake sediments, as a result of the observed effects of urban-based influences entering the lake and the expanding urban development within the lake catchment and greater surrounding Moreton Bay region, regular future measurements of sediments should be undertaken to monitor change in nutrient loadings within Coombabah Lake resulting from increasing urban development.

Analysis of surface sediment samples from 10 locations across the lake found levels for heavy metals were generally below the ANZECC/ARMCANZ (2000) guidelines. Exceptions with Ni and As were found at the upstream reach and entrance of Coombabah Creek.

Typical suboxic to anoxic sediment with extremely low acid volatile levels was measured in lake sediments, and the risk of monosulfides affecting dissolved oxygen levels in the lake is also currently low. It was concluded that there was no evidence of significant risk to the level of dissolved oxygen in lake waters from the resuspension of sediments.

The upstream trends of increasing metal concentrations can be largely explained by natural gradients in the abundance of aluminosilicate minerals in the fine-grain sediment fraction. Suboxic conditions and very low AVS levels reported in sediments, and the strong correlation between HCl-extractable Fe and a range of HCl extractable metals, suggest that reactive Fe minerals such as Fe(III)-oxyhydroxides are major sorbents for As, Cr, Cu, Ni, Pb and Zn. It is suggested that AVS may represent a relatively transient binding phase for trace metals in Coombabah Lake.

Every assessment confirmed that overall the bioavailability risk from metals within Coombabah Lake sediments is currently low. However, the low and generally transient concentrations of AVS present in lake sediments mean that the sediments may not be able to tolerate a significant influx of additional metal sources and hence may not buffer the system sufficiently to prevent a harmful bioaccumulation of metals.

The shell layer found at 0.3–0.5 m is in the order of 2000 years old, suggesting the sediment accumulation in the lake is extremely slow and has not been greatly accelerated by urbanisation. The northern end of the lake does, however, appear to be comprised of tidal delta sands that are actively accreting. This is possibly the result of the separation of the Southport Spit from South Stradbroke Island in 1896 and the creation of the Jumpinpin Bar. This would have accelerated the intensity of tidal exchange in Coombabah Lake, the process being given even greater energy by the creation of the Southport Seaway in 1987. The increasing level of tidal exchange now experienced by the lake may be responsible for its maintaining a well-flushed and healthy sediment and related water quality environment.

Benefits and outcomes

The investigations undertaken with the sediment component of this study all point to the conclusion that the sediments of Coombabah Lake are reasonably healthy and stable. For the community to continue to enjoy the amenity provided by the lake, current levels of management should be maintained.

If future planning decisions allow land-use changes or disturbance to the surrounding wetlands and catchment area, then such developments should be conditioned to ensure the ongoing health of the lake sediments, and sediment quality should be monitored.

Due to the lack of historical data relating to the sediments of Coombabah Lake, it is not possible to assess trends in the health and stability of the region. Therefore future monitoring of the lake should be undertaken to assess any possible degradation in the quality of sediments within the lake environment, using this study's results as baseline data.

Further development

At present and despite a high variability of parameters, Coombabah Lake and upper Coombabah Creek are an example of healthy baseline reference sediment conditions with which to compare other estuarine lakes, bays and streams believed to be affected by urbanisation.

However, as a result of the observed effects of urban influences entering the lake and increasing urban development within the lake catchment and greater surrounding Moreton Bay region, regular future measurements of sediments should be undertaken to monitor for changes in nutrient, metal and sulfide loadings as urban development proceeds.

Chapter 3

Components of sediment transport in the lake

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Chapter summary

This component of the study examines the dominant sediment transport patterns within Coombabah Lake. Particular emphasis is placed on assessing if the wind or tidal currents dominate the sediment transport within the lake. Qualifying the resuspension of bottom sediment in the lake system enables researchers to assess the methodologies required to evaluate such systems in future studies.

To achieve the project aims, a detailed field study was undertaken on the lake over a continuous sample period from 28 October to 14 November 2005. This involved the deployment of eight special instruments capable of recording temperature, water electrical conductivity, water depth and turbidity (the dirtiness of the water) as a function of time. All eight monitoring stations experienced varied periods of tidal inundation. Water samples were also collected manually to determine total suspended sediment loads within the water column.

In addition, a weather station was deployed on a houseboat moored on the lake to monitor atmospheric conditions. The conditions recorded were: air temperature, solar radiation, barometric pressure, relative humidity, wind speed, wind direction and rainfall figures. Continuous data was collected for all parameters at 15-minute intervals from 29 October to 12 November 2005.

The overall results showed that while strong wind events could resuspend sediment from the bed, this was insufficient to permit significant sediment transport capable of removing large quantities of sediment from the lake and into the adjacent Coomera Estuary. Indeed, it appears that only through major floods (not encountered in this study) could sediment be transported out of the lake. Therefore, it is proposed that for the majority of time the lake is infilling, albeit at a very slow rate. Consequently, to maintain the current state of the lake it is essential that the catchment be closely controlled to minimise the influx of terrestrial sediment from overland flow and wind events.

A significant benefit of the current work was the undertaking of a full bathymetric survey, the first undertaken at the site. The highly valuable data collected in this survey will now serve as a benchmark for evaluating future changes and in the design of future studies requiring bathymetric information.

Background

At the end of the last ice age the sea level rose and flooded river valleys, creating coastal bays. Over the last 6000 years the sea level has been relatively constant and these bays have evolved into complex estuaries, through the formation of coastal barrier dunes and infilling by fluvial sedimentation.

Different estuarine systems have now reached different stages of evolution; initial stages of infill result in wave- or tide-dominated estuaries, and later stages can be characterised as wave- or tide-dominated deltas (Harris *et al*. 2002). The stage that an estuary is in can be effectively parameterised on the basis of its morphology, which is an indication of the 'state' of an estuary, and is closely linked to its hydrodynamics and the operative ecological and physical processes (Woodroffe 2003).

Tidal lagoons connected to an estuarine river but having no direct river inflow are a common system encountered in today's environment, including the Coombabah Lake system that forms the focus of this study. In these systems it is common that sediment is carried from a source river channel following floods and deposits on the lagoon bed. Strong wind events that generate waves are often expected to resuspend soft bottom sediment permitting tidal currents to carry some sediment out of—or even into—the lagoon, depending on the wind and tidal conditions. It is expected however that eventually vegetation will colonise the shallows of the lagoon, trapping sediment, reducing wave stirring and ultimately filling the lagoon.

Current state of knowledge about sediment transport processes in shallow estuarine systems

Eisma (1992) describes an estuary as a transition zone from a river to the sea, which is influenced by conditions in the river as well as in the coastal sea. In this environment, suspended sediment can be supplied from a number of sources. Perillo (1995) concurs that the general sedimentology of a specific estuary is the consequence of many conditions, one of the most important being the sediment source (Figure 3-1). This source may be from the river or the adjacent shelf, or transported by littoral currents and introduced into the estuary by tidal action or littoral drift. Erosion of inner estuary bedrock or parent material and biogenic material are characteristic of the particular geological setting of the estuary or the climatic conditions of the estuary.

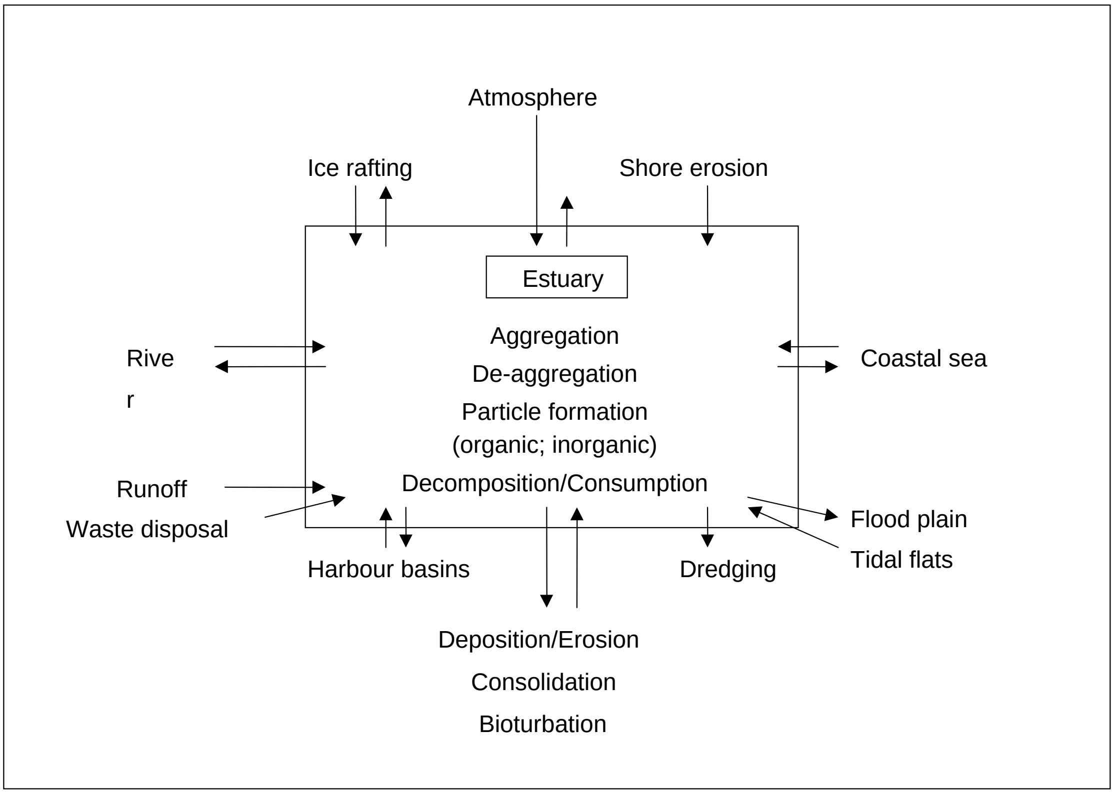


Figure 3-1. Supply and removal of suspended matter in estuaries (Source: Eisma 1992)

Depending on the local setting, one or more of these sources will dominate the supply. Eisma (1992) states that in many tidally mixed estuaries, river supply of sediment dominates the inner part, and supply from the sea the outer part. Tidal mixing may further influence the distribution of sediment, as particles of marine origin may be transported well into the freshwater tidal area.

On the way down the river, continual deposition, re-erosion and transport processes (as shown in Figure 3-2) alter the grain size distribution of the sediment. Much of the coarser sediment can become trapped on the flood plains of the rivers, only being released at times of flood. The finer sediment fractions are transported into the estuary. There the estuarine processes act as a filter on the sediment input, and mixing can take place with sediment brought in from the sea.

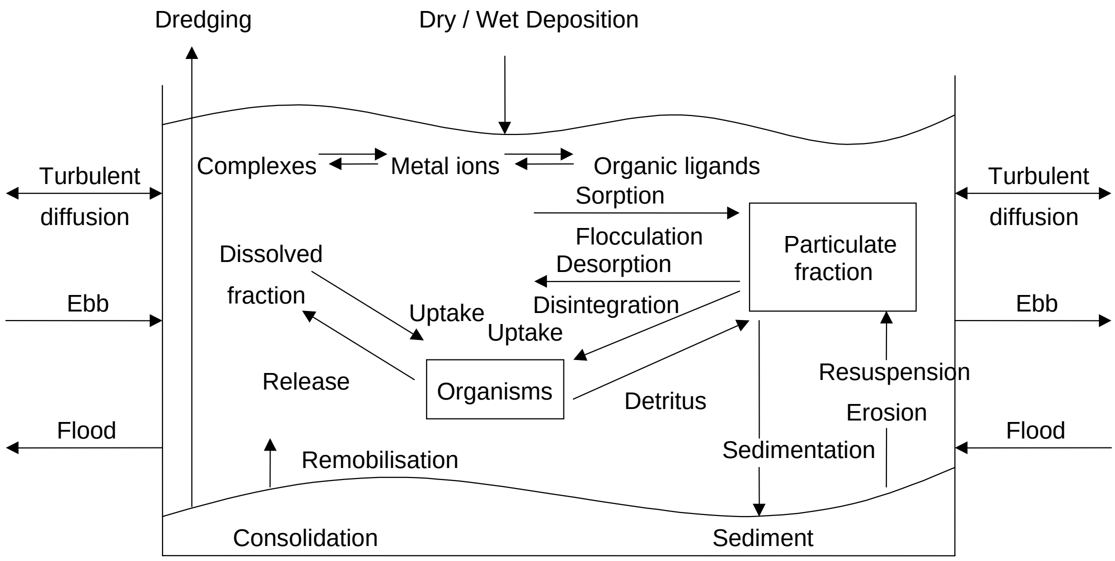


Figure 3-2. Transport processes operating in a tidal river (Source: Michaelis 1990)

Sediments form a crucial control in many estuarine processes. Within the estuary, suspended sediment concentrations are generally high, the particles are fine, cohesive and prone to flocculate, and they are richly organic (Dyer 1995). The energy cycles inherent in the semidiurnal and lunar tides and in seasonal changes cause continual erosion, transport and deposition. Thus, even when little new sediment is coming in, sediments can still silt up harbours and navigation channels.

The tidal elevation characteristics of estuaries create important distinctions in the capability of the currents to move the sediments. Within the estuary the tide can be greatly modified by the friction of the bed on the current, and by the funnelling effect of the convergence of the estuary sides. Dyer (1995) states that a dynamic tidal equilibrium is conceptually possible between the tidal currents and the resulting sediment transport. This implies that the sediment movement causes a change in morphology that alters the tidal current regime, which in turn reduces the sediment transport. The simplest equilibrium concept is that of O'Brien (1969). He found that the cross-section of the mouth of an inlet was related to the tidal prism volume, the volume of the water that has to flow in and out through the mouth on each tide to raise the water level inside. The O'Brien relationship specifies that an increasing tidal prism leads to an increase in the velocity at the mouth of an inlet, which causes the sand to move and the cross-section to increase until the movement diminishes to the threshold value.

Particulate matter in estuaries varies in size, shape, density (composition) and surface characteristics, so that there are large variations in the way particulate matter moves through estuaries. Different types of materials will have different residence times and therefore comprise a different percentage of the total material within an estuary. Within the estuary the transport and deposition of especially the finer-sized particles can be very much influenced by aggregation or de-aggregation, which changes the transport characteristics of the particles so that particulate matter may be removed in a different state from that in which it was supplied. Moreover, the supply from one source will be removed in more than one way (e.g. partly by deposition and partly by outflow) and material removed in one way will have been supplied from various sources. Usually, however, material from different sources cannot be clearly distinguished on the basis of size and composition so that only a gross average residence time or mass balance is calculated involving material of different origin and characteristics (Eisma 1992).

Dyer (1973) states that the sediment transported down a river is generally a heterogeneous mixture reflecting the variety of source grain sizes available within the catchment. However some sorting takes place within the flow as a consequence of the different modes of transport of the finer and coarser material. Michaelis (1990) concurs with this point and expands by stating that sedimentation of suspended particulate matter, resuspension of this material, and possible erosion of consolidated sediments change with the flow and influence the particulate matter concentration in the water column. McDowell and O'Connor (1977) identify three broad groups of material that is transported by water:

1. Particles fine enough to be kept in suspension in pure water by Brownian movement and mutual repulsion due to electrolytic potential. These are clay-sized particles.
2. Particles fine enough to be easily lifted into suspension and transported, largely in suspension, by flowing water. These are silt and fine sand particles.
3. Particles so large that they usually travel by rolling, sliding or hopping between irregularities on the bed such as ripples or dunes. These are the medium and coarse sands and gravels.

Dyer (1995) similarly describes three modes of sediment transport: wash load, suspension and bed load. The distinction between them is not a clear one since there are changes in the grain content of the modes depending on velocity. There are differences, however, between the response of non-cohesive sands and silts and cohesive silts and clays.

Approach and methods

The intensive lake survey study consisted of a continuous sample period between 28 October and 14 November 2005. Sampling was based from a houseboat moored on the lake. Instruments attached to poles driven into the sediment (submersible automated instruments accompanied with data loggers capable of measuring and recording a range of parameters) were deployed within the lake. Additionally, the manual collection of water samples for the determination of total suspended sediment loads within the water column was also undertaken throughout the study period. The direction and strength of currents were also measured at two locations within the lake environment during the intensive field study.

The maximum depth of the lake was expected to be 1.5 m (part of this study was to quantify the lake's bathymetry), with a tidal range of <2 m. The lake is connected with the Coomera River and has no continuous freshwater input, but can and did receive freshwater inflows through rainfall events.

The field trip involved the manual collection of samples in addition to continuous burst-sampling of the water column through the use of submerged instrumentation at eight selected locations or sensor stations within the lake.

Activities

Various tasks were undertaken as part of this field trip. They included:

Bathymetric survey

Surveying of the lake system using a Trimble R7/R8 RTK survey system. Approximately 600 survey data points were collected from within and around the lake system using a stratified block design in order to assess the bathymetric conditions at the time of the survey.

Sensor deployment and automated sampling

The deployment of conductivity, temperature, pressure and turbidity sensors (CTDN probes) to sample water column parameters (through burst-sampling) during the fieldwork period.

Water sample collections

The manual collection of water samples for the determination of total suspended solids (TSS) at each of the sample stations (associated with deployment of CTD sensor stations) to relate the recorded turbidity signal to TSS.

Atmospheric conditions

Weather conditions recorded within the lake system (on research houseboat) using an automated weather station.

Instrumentation used during the study

A variety of instruments were used to undertake the experimental study. The instruments were either set on stakes pushed into the bed, deployed by hand or lowered from a boat. Two boats were used within the lake. These were a houseboat (see Figure 3-3), which acted as the base for the study, and the other was a small tender for moving around the very shallow sections of the lake.

Figure 3-3. The research houseboat used in the study

Note the location of the weather station on the boat (yellow circle) and the mud flats directly behind the boat.

There were four primary instruments used for studying the water column conditions. They included:

Conductivity, temperature, depth and nephelometry (CTDN) probes

These self contained devices, FSI NXIC CTD with internal storage and power are one of the newest instruments on the market. They were ideally suited to the task. They were used to record temperature, conductivity, pressure (depth) and nephelometry as a function of time. Details of this instrument can be found at <www.falmouth.com>.

Nortek Aquadopp acoustic Doppler current meter (ADCP)

Two ADCPs were used to record the flow velocity data at the mouth and upper reach of the lake. These were 2 MHz Nortek Aquadopps. The Aquadopp® profiler measures the current profile in water using acoustic Doppler technology. It is designed for stationary applications and uses three acoustic beams slanted at 25° to accurately measure the current profile. Further details can be found at <www.nortek-as.com>.

Trimble RTK GPS

The bathymetry of the lake and topography of the surrounds were determined using an RTK survey system. This consisted of a Trimble R8 base station and R7 roving unit. Further details of the equipment are available at <http://www.trimble.com/productsaz.shtml>.

Weather station

Environdata's WeatherMaster 2000 weather station was used to record atmospheric conditions on the lake. This is a complete, self-contained, solar-powered unit that incorporates a purpose-built data logger with the six most commonly used sensors: rainfall, air temperature, relative humidity, wind speed, wind direction and solar radiation. Further details can be found at <http://www.environdata.com.au/environdata/Weather_Stations.html>.

Within-lake study stations

Eight sensor stations were deployed within the lake. Their locations and the instruments associated with them are presented in Table 3-1.

Figure 3-4 shows the approximate locations of the instruments (along with the weather station mounted on the houseboat). The houseboat was positioned at the green circle location (near Station 8) in Figure 3-4 for 29 October 2005 then moved to the red circle location (closer to Station 4) for the period between 30 October and 3 November 2005. The research vessel was then returned to the green circle location in safer (deeper) waters for the remainder of the study survey period.

Table 3-1. Sensor stations and instrumentation within Coombabah Lake

Station no.	Location	Instrumentation	Elevation from bed (m)
1	27°54.851 S 153°20.893 E	A. CTDN B. 3-D ADCP	0.15 0.30
2	27°54.720 S 153°20.201 E	A. CTDN	0.15
3	27°54.528 S 153°20.736 E	A. CTDN	0.15
4	27°54.497 S 153°21.177 E	A. CTDN	0.15
5	27°54.533 S 153°21.382 E	A. CTDN	0.15
6	27°54.322 S 153°20.828 E	A. CTDN	0.15
7	27°54.109 S 153°20.968 E	A. CTDN	0.15
8	27°54.289 S 153°21.470 E	A. CTDN B. 3-D ADCP	0.15 0.30

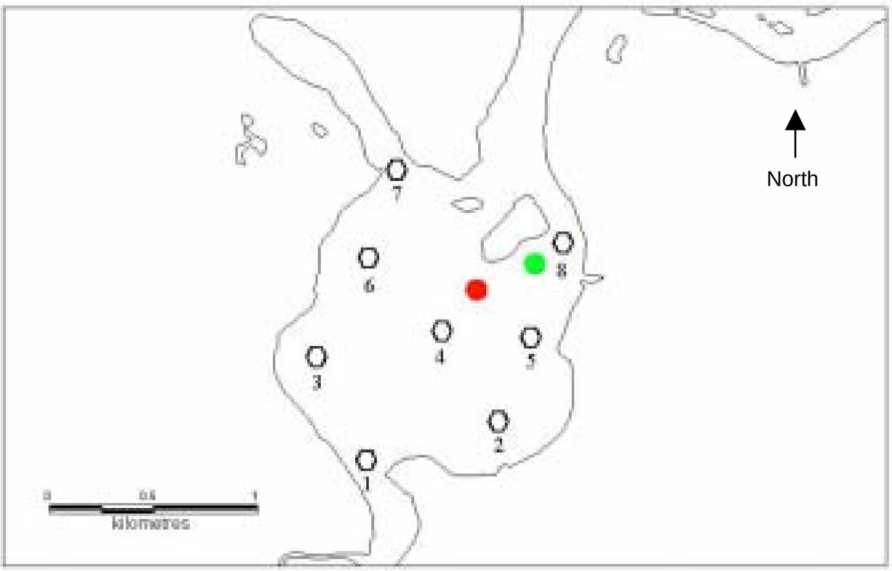


Figure 3-4. Approximate deployment positions of submersible automated instruments (numbered 1–8) and houseboat sampling locations (circles)

All eight monitoring stations experienced varied periods of tidal inundation:

Station 1 was subjected to a maximum fetch of ~1700 m in northerly winds and ~400 m in southerly winds. Surface sediments here comprised predominantly silts and clays. Instrumentation remained continually inundated, with water depths approaching 1.75 m during high tide periods.

Station 2 was subjected to smaller fetches than Station 1 in both northerly and southerly winds (maximums ~1100 and 200 m respectively). Westerly winds provided an approximate fetch of 1000 m. During periods of low tide the Station 2 instrumentation became exposed to the atmosphere.

Station 3 was situated to receive wind waves from the most predominantly occurring winds within the region (southeast and northeast). Fetch distances were ~1200 and 1400 m for southeast and northeast winds respectively. Surface sediment grain sizes surrounding Stations 2 and 3 were characterised by a similar trend to station 1, with sediments dominated by muds (< 63 µm).

Station 4 was located within the central area of the lake and as such was affected by winds from all directions with similar fetch distances (~600–800 m). Instrumentation remained submerged during the entire sample period. Surface sediments were dominated by sands (> 63 µm).

Station 5 was located in the northeastern section of the lake which experiences reduced wave activity as a result of the limited fetch in southeasterly and northerly winds and the limited occurrence of westerly winds. The sensor station was located in a shallow region of the lake and as a result the instrumentation remained exposed for extended durations during periods of low tide.

Station 6 was characterised by large fetches for both southeasterly (~1200 m) and northeasterly (~1000 m) winds, with a very limited fetch under westerly wind conditions. Unlike Station 3, however, Station 6 exhibited surface sediments dominated by fine sands (> 63 µm). Station instrumentation became exposed during periods of low tide.

Station 7 was subjected to a maximum fetch of ~1400 m in southeasterly winds. Surface sediments comprised predominantly silts and clays. The sensor station became exposed to atmospheric conditions for extended periods of time during periods of low tide.

Station 8 was located at the northern lake entrance. The surrounding shore configuration is such that the fetch during winds from any direction other than south and southwest is insignificant. The instrumentation remained continually inundated, with water depths approaching 1.75 m during high tide periods. Surface sediments were dominated by fine sands (> 63 µm).

At each sample station location, extended star pickets with attached instruments were driven into the lake floor to provide a fixed structure from which continual burst measurements could be made.

Instruments were deployed simultaneously for a period of 11 days during the period 30 October to 10 November 2005. Conductivity, temperature, depth and turbidity high-frequency data were collected using time-averaged data (3.5 min bursts at a frequency of 10 Hz, Stations 1–3; and 5–8 min bursts at 5 Hz, Station 4) obtained every 15 min. ADCP data was collected for 22 min at a wave interval of 30 min and a sample frequency of 1 Hz (profile interval of 60 sec).

Meteorological measurements

Wherever possible, including for bodies of water such as lakes, wind direction and wind speed should be directly measured at the study location. Therefore, for the present study, meteorological data on the wind direction, speed and duration at the study location was measured within the lake environment.

During the study, atmospheric conditions were continually monitored around 5 m above the lake surface using a weather station (WeatherMaster 2000 ®; Environdata) mounted on the roof of the stationary base research vessel within the lake in an area that was not sheltered by topography. Continuous data was collected for all parameters at 15 min periods from 29 October to 12 November 2005.

Topographic surveys

The meso-scale topography of the entire lake system and adjoining creeks was surveyed using a wireless roving global positioning satellite (GPS) system attached to a fixed length (2 or 3 m) pole sampler (R8 GPS receiver; Trimble Navigation Inc.) in association with a fixed base station (R7 GPS system; Trimble Navigation Inc.). From a research vessel or shore, the sample pole on which the R8 GPS receiver was positioned was placed onto the lake floor at 578 sample points in a stratified block design approach over a 3-day period during high tide phases to determine the lake system bathymetry. The survey instrument measured topographic height points with a typical accuracy of 5 cm. The base station (R7 unit) was positioned within 2 km of the lake with all data referenced back to the Gold Coast City Council reference point 116004.

A significant limitation of the surveys was the inability to undertake topographic surveys within the forests surrounding the lake. This was the result of canopy restrictions on the use of the roving unit, as the branches and leaves blocked the satellite signals.

Suspended sediment concentrations

In addition to the CTDN probes, water samples were collected throughout the study so that the raw CTDN nephelometric signals could be converted into suspended sediment loadings. When 250 ml water samples were collected, turbidity was also recorded on-site using a nephelometric turbidity meter (Analite 160 turbidity meter; McVan Instruments).

Results and discussion

Bathymetric survey

A major activity of this study was to undertake a detailed bathymetric survey of Coombabah Lake and its immediate surrounds. Such detail was not previously available and thus it was not possible to develop any quantitative understanding of the rate of change of the bed over long time scales (order of years).

From an RTK survey conducted during the field experiment, the elevation data were corrected to Australian Geocentric Data Zone 56 J (horizontal data) and then plotted to reveal details of the bottom bathymetry relative to the Mean Water Level (=Australian Height Datum [AHD]–0.09 m, vertical data). The results of this are presented in Figure 3-5. The small crosses on the model show the locations of the survey points and it is important to note that only elevations contained within the bounds of these points should be considered valid. The plot was generated using the commercial software SURFER with the Kriging contouring method. Survey points were not collected within the fringing forests as the canopy leaves and branches prevented the collection of topographic data.

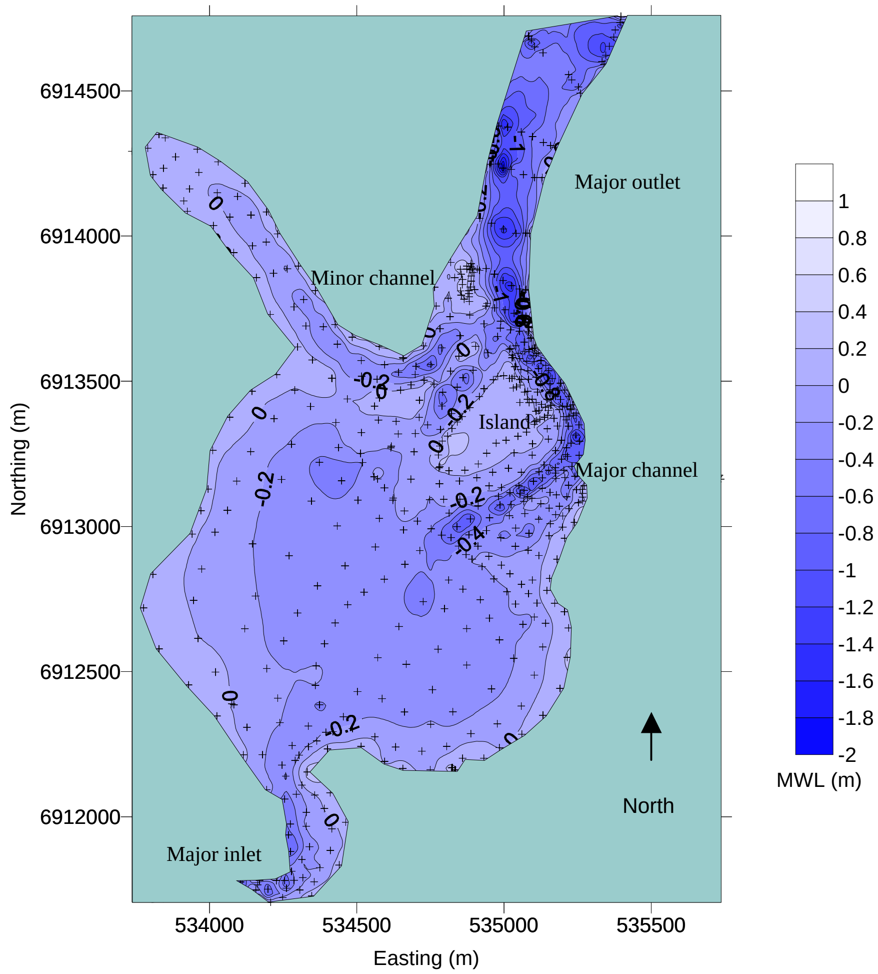


Figure 3-5. Digital elevation model of Coombabah Lake

The small crosses show the locations of the survey points.

Figure 3-5 shows the main channel through the lake extending from the primary major inlet to the outlet. What was of interest was the mapping of the expected major channel and the locating of a minor channel to the west. These two channels indicated that tidal flow could maintain some form of channel in their area of confined flow. Overall, the lake was found to be very shallow and predominantly exposed at low tide.

Weather conditions encountered during the study

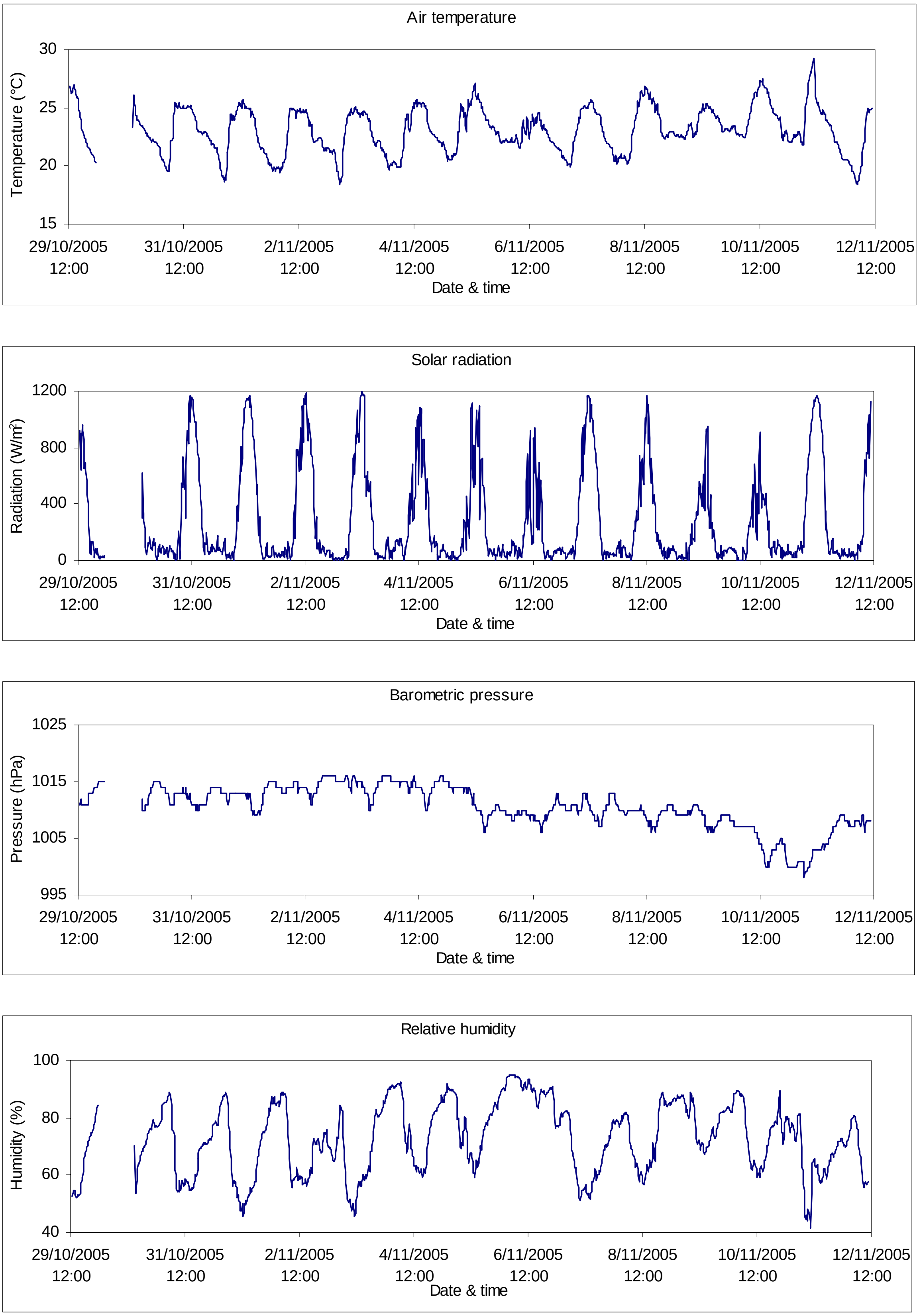
The weather conditions encountered during the study were of varied nature. Plots of the collected data are presented in Figures 3-6a and 3-6b.

Air temperatures were mild and ranged from 19 to 30 ºC, while the solar radiation revealed variations in cloud cover during the day. The air pressure showed a general decrease in pressure with time, with regular fluctuations (such as frontal passings).

Wind conditions varied significantly during the study. They were generally calm during the night, with maximums encountered in the early afternoon. Wind speeds reached a peak of 30 km/hour, while the directions for the strongest winds were generally from the southeast and northwest (the dominant wind directions for the coastal southeast Queensland region according to the Bureau of Meteorology—see wind rose data available at: <http://www.bom.gov.au/climate/averages/wind/selection_map.shtml>). Further comparison with data available from the Australian Bureau of Meteorology for Southport (located just to the south of Coombabah Lake, latitude 27°98’ S, longitude: 153°41’ E) revealed that the measured wind speed patterns were typical for this region, albeit at a reduced speed due to topographic sheltering and increased frictional effects. Therefore, the lake’s *normal conditions* were encountered during the study, but without extreme events such as cyclonic storms and major flood events (which are expected to have dramatic influences on all aspects of the region).

It is also noteworthy that the region encountered rain events during the study period. The events were found to be very isolated and not always recorded by the rain gauge.

The lack of extreme events encountered in the study program is typical for any study of this type (even those undertaken over a year) and reveals an expected shortcoming. That is, they are limited in possible events encountered and interpretations based upon scientific judgement must therefore be made. However, as this study was undertaken during typical wind and hydrodynamics conditions, it forms a unique baseline data set that can be used in future studies.



3-6a. Atmospheric weather conditions encountered during the study: air temperature, solar radiation, barometric pressure and relative humidity

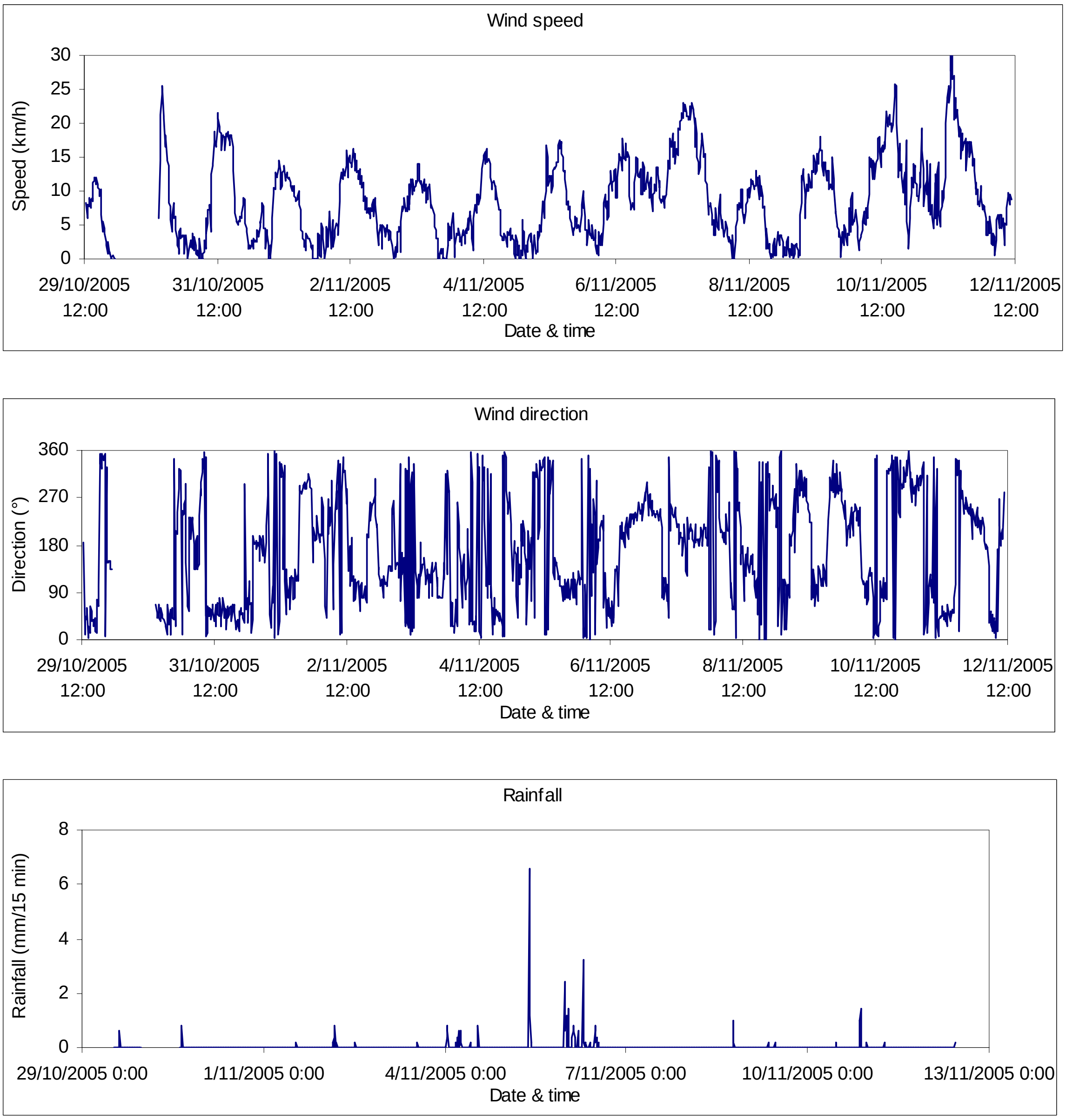


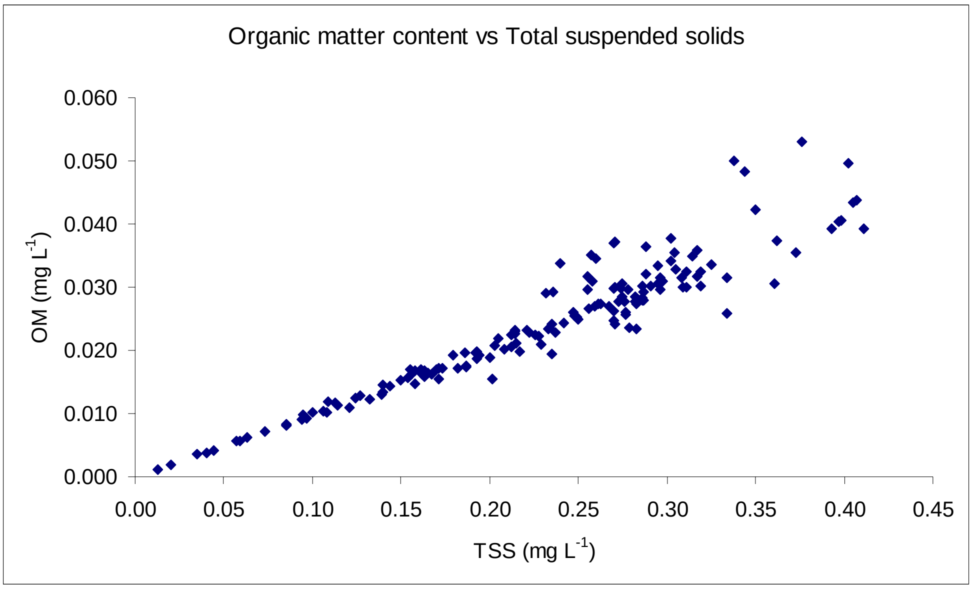
Figure 3-6b. Atmospheric weather conditions encountered during the study: wind speed, wind direction and rainfall

Suspended sediment property measurements

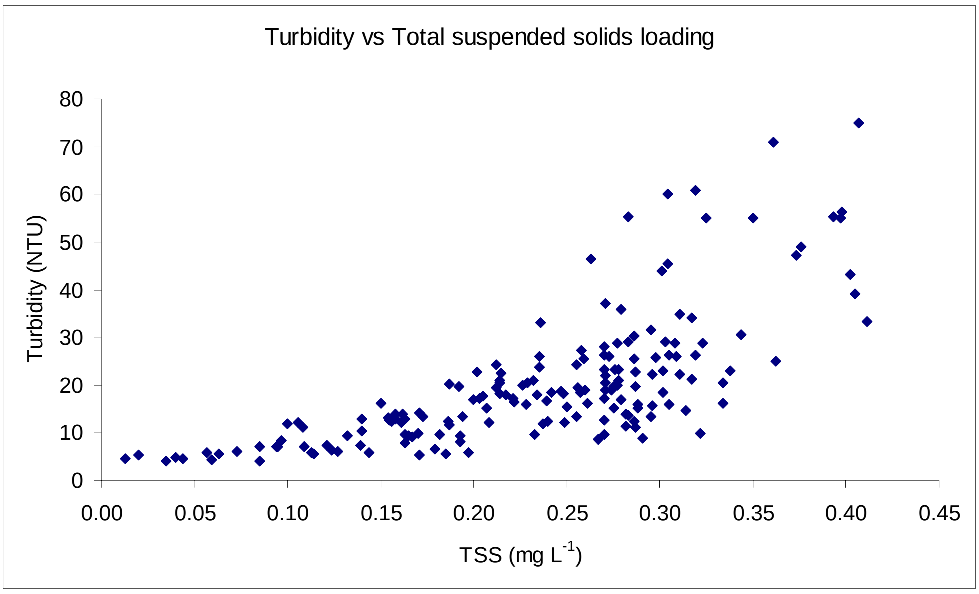
As the direct measurement of suspended sediment loads is a difficult and extremely time-consuming activity using gravimetric analysis, turbidity sensors are often used for the long-term monitoring of suspended sediment loads. These instruments (as used here – CTDN probes) are deployed and turbidity data, along with other parameters, collected on a regular basis (here every 15 min). Additionally, water samples are manually collected and the suspended sediment loads assessed through gravimetric analysis (here approximately one site per tidal cycle). By collecting the data at similar times the results can be used to calibrate the instruments' turbidity data signal to give suspended sediment loads.

In addition to the determination of total suspended solids (TSS), further analysis of the water samples collected through the manual sampling process can be used to determine the organic (combustible) and inorganic (non-combustible) components through standard techniques. This process essentially involves ashing the material remaining on a filter paper (at 500°C) so that the organic material is burned off. The difference in weight between the pre- and post-ashed sample represents the organic component. For this study the results (see Figure 3-7a) showed that the organic matter fraction of the TSS was approximately 10%, giving an inorganic component of approximately 90%. The organic matter fraction within the lake during the sampling periods remained relatively unchanged under different conditions, except at high turbidity levels where the organic fraction was significantly higher.

As shown in Figure 3-7b, the comparison between the collected instrument data and the gravimetric analysis revealed a linear relationship for all sites for low-to-moderate suspended sediment load realisations (<200 mg L^{-1}), indicating the sediment is of similar type. However, for high TSS levels (>200 mg L^{-1}) the scatter was significantly greater, indicating suspended sediments of different types were being sampled. The high variations located at high TSS levels are associated with inflow events from the main inlet following catchment rain events. The exact reasons for this are unclear. These results indicate that further data, including particle size distribution and chemical composition, would need to be collected in future studies in order to ensure that such variations can be readily explained.



(a)



(b)

Figure 3-7. (a) Plot of organic matter versus total suspended solids (TSS) and (b) plot of TSS load versus turbidity (NTU) as collected with the Analite 160 turbidity meter for all stations

Sediment distribution patterns

Major trends

The deployment of the CTDN sensors permitted the determination of turbidity levels throughout the lake as a function of space and time at the eight fixed locations.

Figure 3-8 shows the data collected by the CTDN sensor located near the mouth of the major outlet channel (Station 8). The figure shows how the water level at the site varied with time as a result of the tidal fluctuations within the lake (the region is dominated by semidiurnal movement). Similar fluctuations were seen in the conductivity and temperature signals as water moved into and out of the lake with the tides. The turbidity levels also were generally low (~50 NTU) and observed to fluctuate with the tides. The major exception to this was on the afternoon of 6 November 2005 where the levels rose to 450 NTU. Examination of the other records indicates that this elevated signal was likely caused by biological contamination (i.e. it was a sustained isolated event not evident at other sites).

Figure 3-9 shows data collected within the central region of the lake (Station 4). Compared with Station 8’s, the Station 4 data shows a similar nature in tides, temperature and conductivity, further indicating the exchange of water with the adjacent estuary. The turbidity levels here were slightly higher than at Station 8, with no significant variations other than those associated with tidal fluctuations.

Figure 3-10 shows the data collected by the CTDN sensor located near the mouth of the major inlet creek (Station 1). Again the pressure, temperature and conductivity signals fluctuated with tides. However, it is evident that other processes must have been encountered as the turbidity levels were highly elevated compared to the other sites, and with greater fluctuations. It is noteworthy that the levels were so elevated at times that the instrument railed and could not make correct readings. As this site is located near the major inlet creek it is also an area subjected to the impact of rainfall events within the catchment. That is, there is the likelihood of greater terrestrial sediment inflow. This certainly appears to be the case as rainfall events were encountered during the monitoring period.

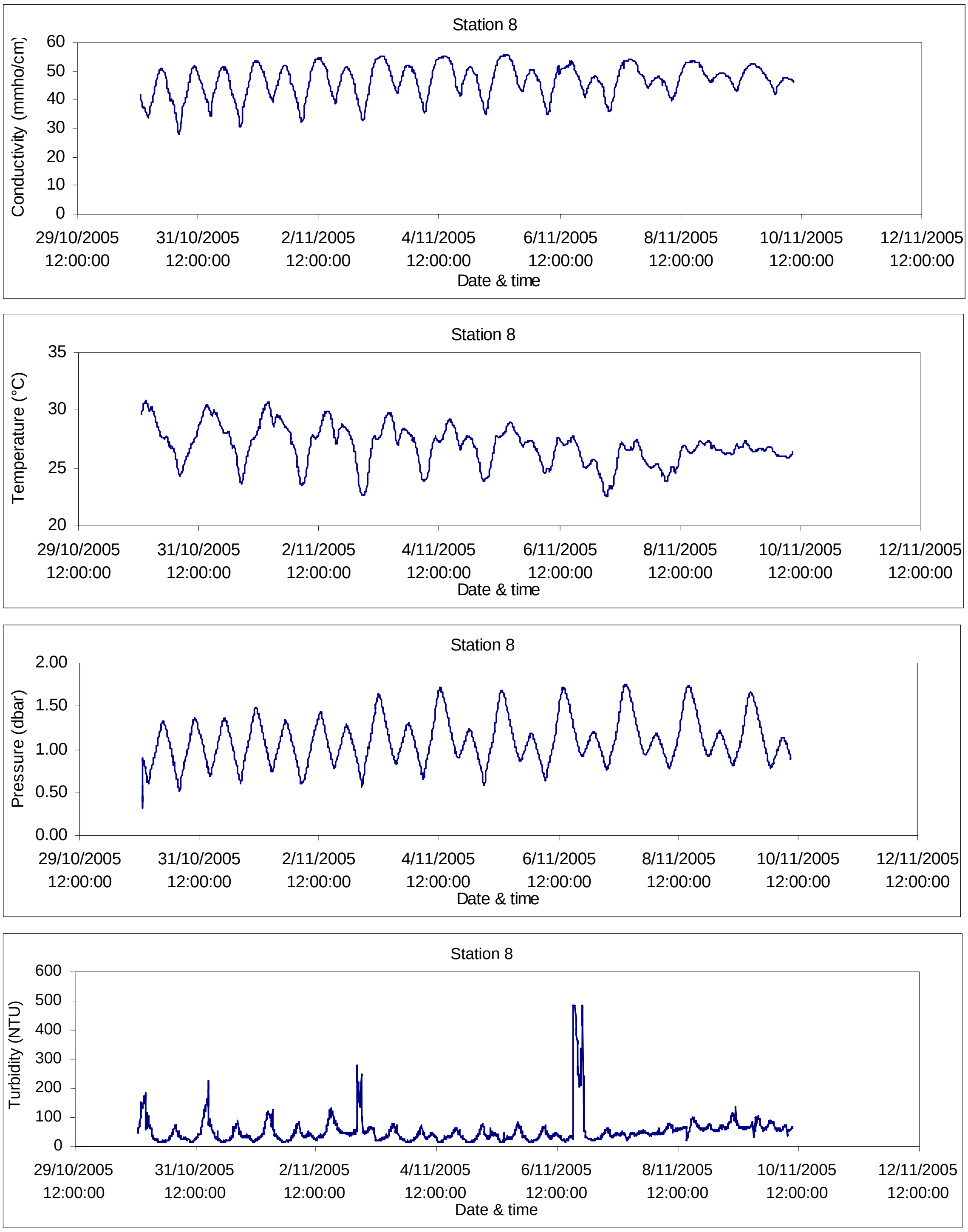


Figure 3-8. CTDN measurements made at Station 8, near the mouth of the outlet channel (Note pressure is relative to the CTDN probe)

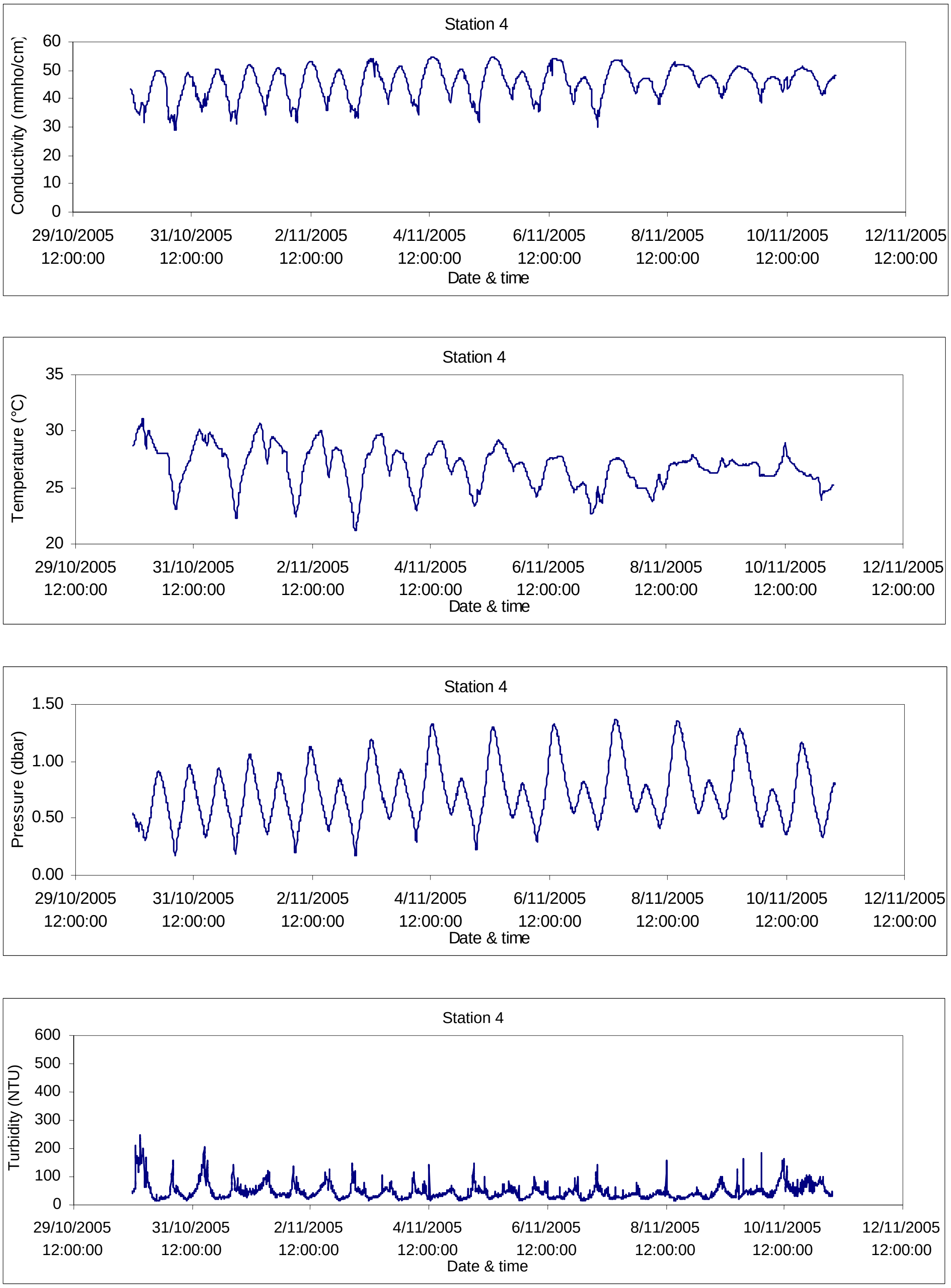


Figure 3-9. CTDN measurements made at Station 4, near the centre of the lake (Note pressure is relative to the CTDN probe)

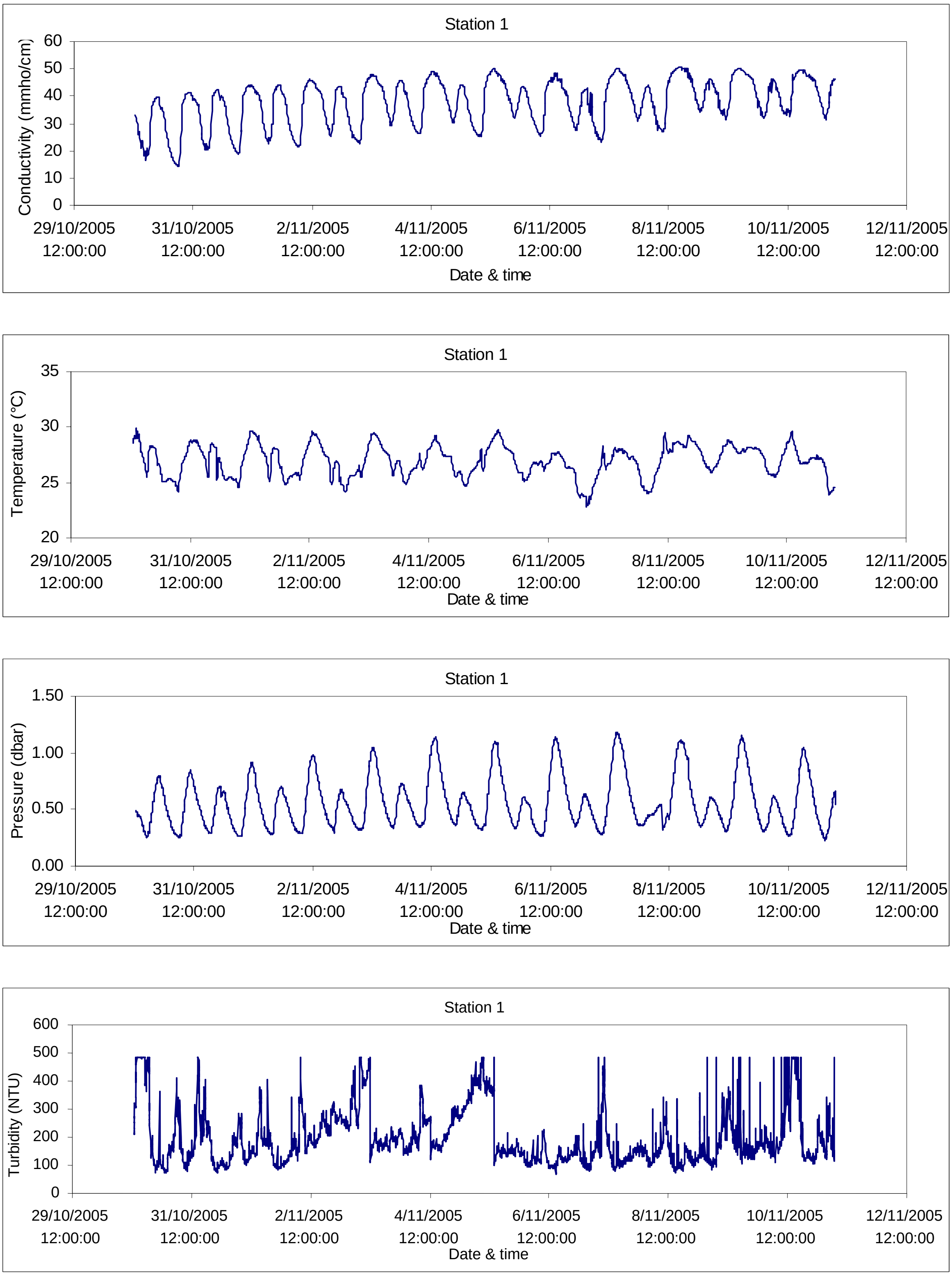


Figure 3-10. CTDN measurements made at Station 1, near the mouth of the main inlet creek (Note pressure is relative to the CTDN probe)

From the data presented in Figures 3-8, 3-9 and 3-10 it is evident that there is a gradual decrease in turbidity levels from the major inlet to the major outlet. This indicates that as sediment enters the lake through the inlet it settles onto the lake bed and is trapped. The wind events encountered were unable to resuspend significant sediment loads for them to be transported out of the lake, indicating that the lake would act as a sink under typical conditions. The only exception to this would be during major flood events where major losses to the lake bed are expected as a result of increased bottom stresses associated with floods.

Figures 3-11a and 3-11b show turbidity levels from the remaining within-lake stations (Stations 2, 3, 5, 6 and 7), that is, further from the inlet and outlet points than Stations 1 and 8. Large spikes (short-lived elevated events) are likely to be the result of temporary contamination and not reflective of typical turbidity conditions. The regions of no signal represent times when the instruments were out of the water.

Another feature determined from the tidal data is that the tidal lag increases with distance up the lake (see Chapter 4 for additional details) and that there was significant tidal asymmetry. The flooding tidal period was approximately 5.5 hours while the ebbing tide was approximately 6.5 hours. The consequence of this is that flooding velocities were greater than ebbing velocities, with the net result being the upstream pumping of resuspendable sediment (Eisma 1992). This also supports the notion that the lake will act as a sediment sink, as any resuspended material will be moved upstream to regions of low flow (away from the channels) where it can settle, except of course during major flood events where floodwaters will cause sediment to be transported out of the lake.

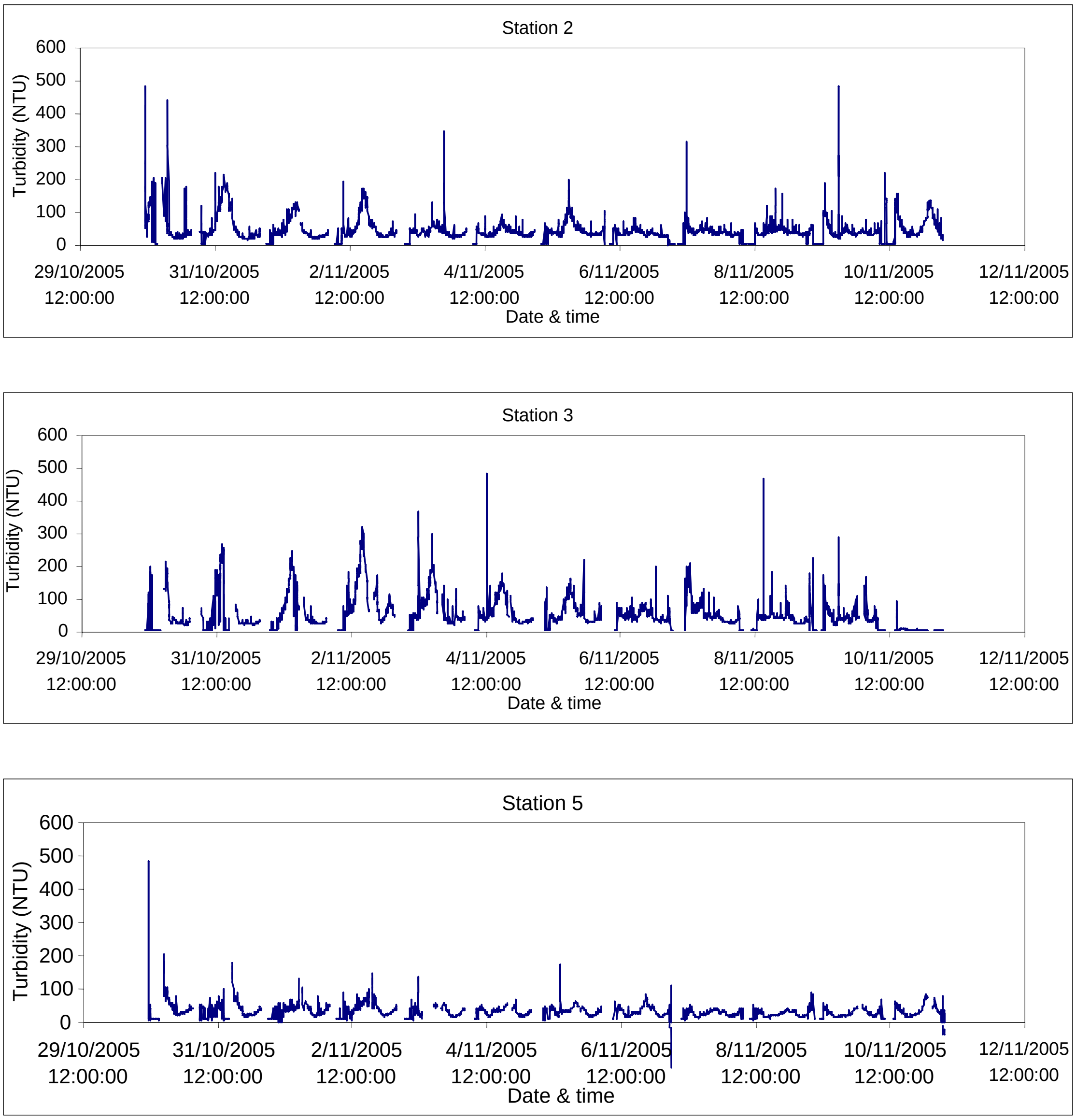
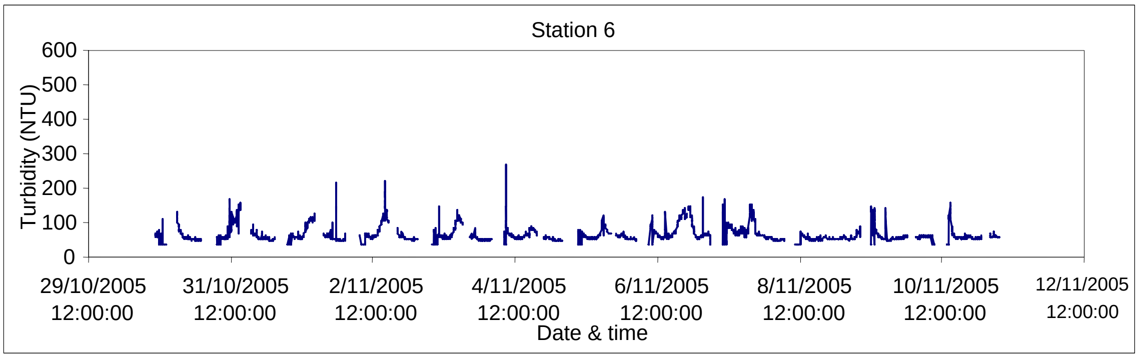


Figure 3-11a. Turbidity level readings from Stations 2, 3 and 5



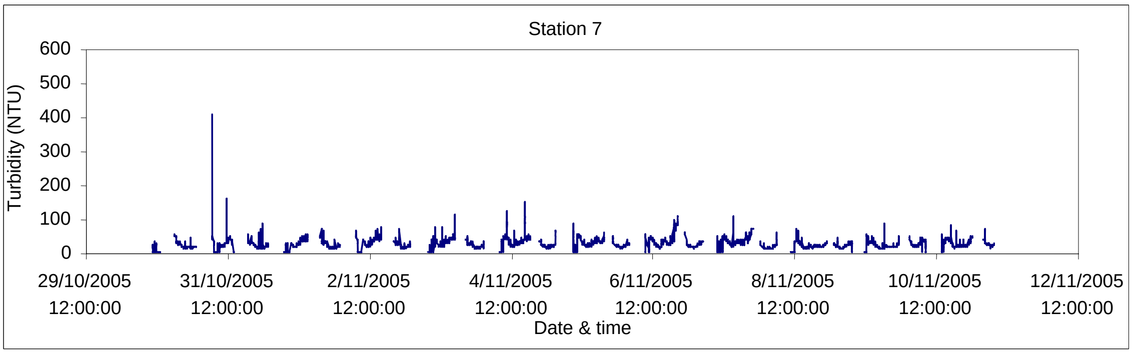


Figure 3-11b. Turbidity level readings from Stations 6 and 7

Methodological issues—what constitutes appropriate sampling?

One of the main aims of this study was to consider what constitutes appropriate sampling in order to assess the sediment resuspension dynamics within a shallow lake. For this study eight CTDN instruments were deployed within the lake. This is a highly costly exercise as each CTDN is valued at over $12 000 and requires batteries for each study period ($200 per CTDN per 2-week deployment). Furthermore, apart from when they are positioned in areas of restricted access, these instruments are subject to the possibility of theft unless they are under constant surveillance.

When all the turbidity data is examined, it is evident that there are variations throughout the lake and thus as many sensors as possible are therefore required to build up a detailed picture of the overall lake suspended sediment dynamics. The box plots shown in Figure 3-12 reveal the variations encountered across all sensor stations.

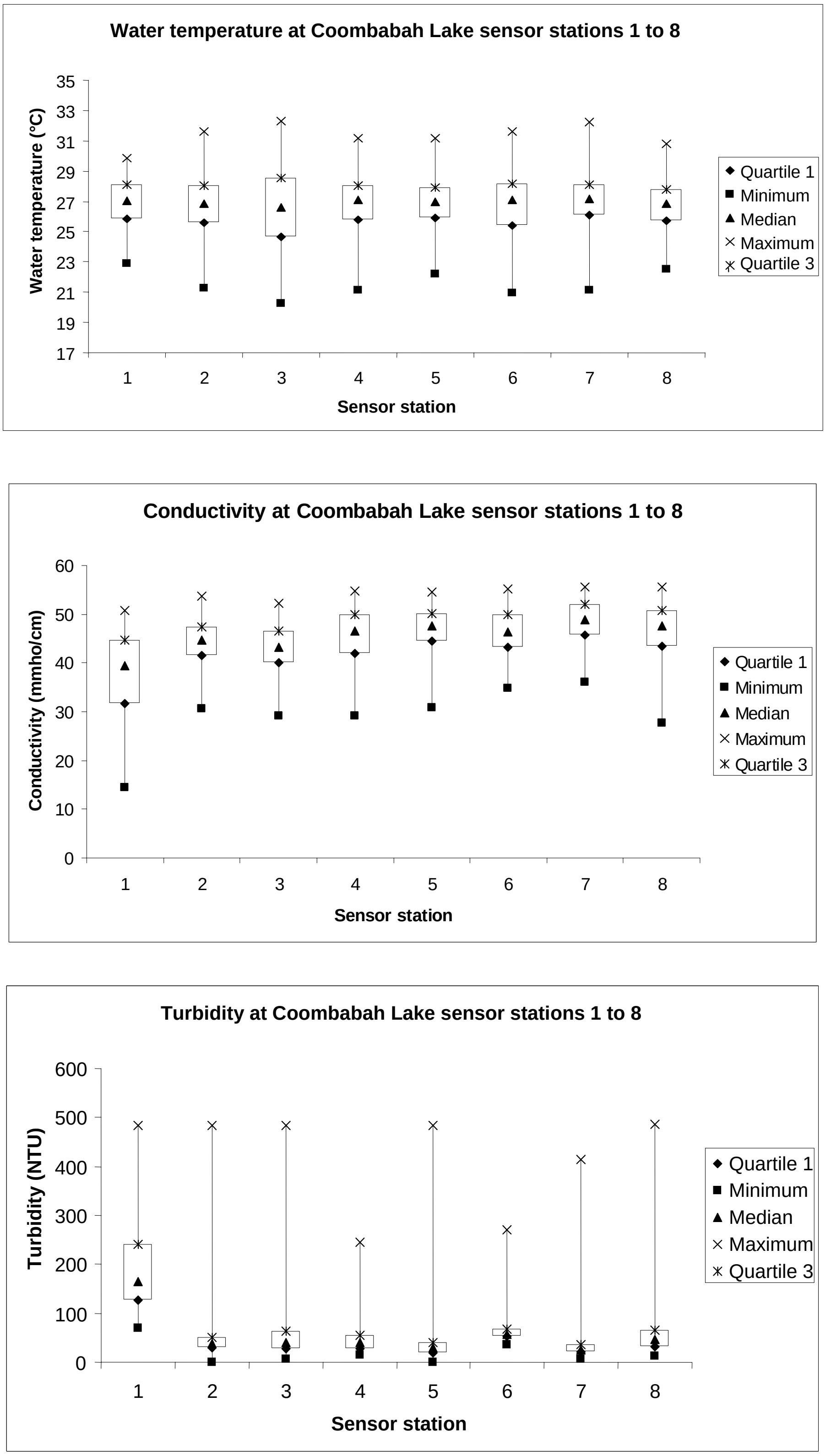


Figure 3-12. Box and whisker plots of comparative temperature, conductivity and turbidity readings across all stations

Plots show the median, the interquartile ranges (1 and 3) and the maximum and minimum data values. For turbidity the upper quartile was influenced by the sensors being temporarily contaminated.

The data in Figure 3-12 shows that temperature variations are uniform throughout the lake about a relative uniform median value but conductivity varies more significantly, primarily due to the influx of fresh water from the major inlet adjacent to Station 1. The greatest variations are observed in the turbidity data. The highest values encountered are associated with high sediment loadings at Station 1 that followed inflow events and apparent temporary sensor contamination (large narrow spikes). The maximum NTU level that can be detected by the CTDN probes is 480 NTU, with any levels greater than this being electronically limited to 480 NTU. The temporary contamination (e.g. just after midnight on 4 November 2005 at Station 3, see Figure 3-11a) was likely the result of biological contamination (such as a small crab moving on and off the NTU sensor). This could be deduced as the events remained short-lived, with the instruments rapidly returning to background type conditions. This shows a limitation of this type of study in that contamination can influence the results. To help alleviate this, the instruments deployed here were cleaned daily.

The data again reveals that the wind did not necessarily play a major role in developing sediment resuspension (i.e. there was a poor relationship between wind and turbidity levels), but that advection (as seen through the relationship between water elevation and turbidity levels) was dominant. This advected material would most likely have been ‘unsettlable’ material (i.e. material that does not settle over a tidal cycle). It is also evident that the material entering the lake from the main inlet creek (Station 8) rapidly settled onto the bed (as there was a constant reduction in average turbidity from Station 8 to Station 1).

The sampling rate chosen for this study appears to have been appropriate. It has captured events and permitted the determination of lake conditions during the study. More sensors would have permitted a greater coverage of the lake, while more time on site would have permitted the collection of a longer data set. Further instruments would have been valuable in determining the settling behaviour of the sediment from the creek—that is, in hindsight it would have been ideal to have more instruments placed near the inlet creek mouth.

Overall, the sampling was appropriate for determining the general behaviour of the lake, but additional time and instruments would have permitted greater detail to be determined. The methodology chosen (given instrument and financial limitations) was appropriate for gaining background data and highlighting areas that may require further investigation.

Conclusions

This study employed an intensive, short-tem data collection process to examine the sediment dynamics within Coombabah Lake. The study focused on attempting to assess the relative importance that wind, tides and sediment settling play on the sediment dynamics within the lake. For the study period, it was observed that once sediment entered the lake from the major inlet creek, it rapidly settled on the bed where it was locked away from future movement (except in the event of extreme events). The implication of this is that should there be an increase in sediment influx into the lake (e.g. due to a change in urbanisation patterns within the catchment), the lake may infill, resulting in a change of environment.

The highly valuable bathymetric data collected in this study can now be used as a base map for evaluating future changes and in the design of future studies requiring bathymetric information.

Benefits and outcomes

CRC partners and stakeholders benefit from this project as it has identified important sediment transport mechanisms within a shallow, subtropical lake which is likely to be representative of other such systems of similar character. It has found that sediment is advected through the lake by tidal flows and that while wind events may cause some resuspension, wind does not appear to be a dominant sediment transport initiation mechanism. For the community to continue to enjoy the conditions within the lake, the current levels of management should be maintained.

As previously mentioned, another significant benefit to stakeholders and to the scientific community was the undertaking of the full bathymetric survey (with a vertical resolution of ~5 cm), the first undertaken at the site. The survey highlighted the need for gaining such essential detail so that potentially important regions can be identified before a hydrodynamic study commences.

Further development

The study has also shown that the highly complex nature of such systems as Coombabah Lake requires extensive monitoring if a detailed understanding of its physical behaviour is to be determined, and that to be able to predict the impact of future development in the area a full numerical predictive model is required. This model could be calibrated using the extensive, but temporally limited, data set collected.

A predictive model should consider the hydrodynamics of the lake, the sediment types within the lake and, importantly, the likely change in sediment types entering the lake as a result of the expanding urban development within the lake catchment and greater surrounding Moreton Bay region. Regular future measurements of bathymetry and bed and inflow sediments should be undertaken to monitor for trends and changes in the region as urban development proceeds, so that any problems that may arise can be acted upon rapidly.

Chapter 4

Patterns of tidal flushing at the mangrove fringe of the lake

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Chapter summary

This chapter reports on the patterns of tidal flooding at the mangrove fringe of Coombabah Lake. The area is urbanising rapidly and pressures on wetlands are increasing. The role of mangrove systems in the larger context was not well understood—for example their contribution to the habitats of other organisms such as fish and crabs. As well, there are negative issues with the wetlands as they provide habitats for disease-bearing mosquitoes. Ross River and Barmah Forest viruses are the major ones in the Coombabah Lake area. The mangroves thus play a dual role: in providing important habitat for wildlife and in providing habitat for mosquitoes. The nature of the flooding pattern and its size and duration are key elements for the mosquito species in this area, which require a variety of water levels for their survival.

These investigations originally set out to quantify the flow patterns within Coombabah Lake and those connecting the lake with the surrounding mangrove ecosystems. The aim of the research was thus to identify the detailed surface hydrology for the range of forest types and connected mangrove forest areas around the lake. It became obvious that the system was more complex than originally believed and so tidal observations—both contemporary and long-term—at a nearby official tidal station (referred to as the 'standard port') were analysed from a variety of perspectives. Detailed local observations described in this chapter were related to the broader, long-term analysis.

Three sets of analyses were undertaken:

1. *Tidal characteristics of the standard port*
   This provided a detailed description of the tidal characteristics of the nearby Gold Coast tidal station, including analysis of long-term (19-year) tidal observations.
2. *Within-lake tidal observations*
   This part of the study examined tidal observations from tide gauges deployed within Coombabah Lake and Coombabah Creek for which the full extent of tidal activity was observed unimpeded.
3. *Lake-edge tidal observations*
   The third and final analysis examined tidal inundation (surface flooding) of lake-edge mangroves by deploying tide gauges at a range of locations in the mangroves. These data were related back to the tidal observations for the standard port in order to estimate the 'trigger' height at which tidal flooding in the mangroves commenced. This analysis provided an

overview of Coombabah Lake’s lake-edge mangrove forest hydrology as well as a detailed picture of how different parts of the system can be expected to function under a range of hydrologic conditions.

Exploring flooding patterns allowed researchers to determine if high tides flooded all mangrove areas adjacent to Coombabah Lake and what kind of water fluctuations occurred, as a basis for informing other aspects of urban wetland understanding.

An average timelag of almost three hours was observed between high tides at the standard port and high tides in Coombabah Lake. Flooding was not even over the study area but patterns were identified in areas that flooded at different ‘trigger’ tide heights and with different durations of flooding. Based on observed tidal characteristics, there appear to be two distinct forms of basin forest: those inundated with relatively deep water (around 40–50 cm) and with relatively infrequent exposure to tidally driven flooding; and those with relatively shallow water (around 10–20 cm) with relatively frequent exposure to tidal flooding.

Overall, the main two points to be derived from the results are: (i) that the mangrove system at Coombabah Lake is not always flushed by tides; and (ii) that the very distinct hydroperiod characteristics observed at the study site are related to tidal elevation and substrate form. These findings have implications, firstly, in light of sea level rise and other possible consequences of climate change, and secondly in relation to the effects of urbanisation and wetland management on the area generally.

Background

The Coombabah Lake research

Research in the general field of mangrove hydrology in the Coombabah Lake region began in 1995, prior to the current study. It took the form of a relatively small, focused project to assist mosquito control operations at the Gold Coast by identifying those places within the mangrove forest system that were significant mosquito larval habitats.

This early work unearthed some important questions about connectivity both within the mangrove system and between the mangrove system and Coombabah Lake. As a result, this component of the current study considered the wider implications of mangrove microtopography and tidal influences to inform other parts of the overall urban wetlands body of knowledge and ultimately to help satisfy the need for sustainable management of the system, both to conserve its ecological values and to minimise any adverse impacts of activities related to urbanisation.

General research on mangrove system hydrology

Mangrove forests occupy zones of transition between terrestrial and marine ecosystems. It is generally understood that intertidal wetland characteristics are determined by the cumulative and complex interactions between hydrology, landscape position, sediment dynamics, storm-driven processes, sea level change, subsidence, and colonisation and disturbance by animals (Varnell *et al.* 2003). The degree of influence that each of these factors exhibits determines the structure and function of wetland systems but hydrology is the driving variable (Varnell *et al.* 2003). The importance of hydrology is also noted by Mitsch and Gosselink (1993) and specifically for mangroves by Lewis III (2005). Much of the literature on intertidal hydrology and associated factors tends to focus on salt marsh (e.g. Nuttle & Harvey 1995; Hughes *et al.* 1998; Varnell *et al.* 2003). This may be attributed in part at least to the difficulty of surveying the mangrove systems to collect baseline data.

Mangroves play an important role providing habitat for marine organisms and this is recognised by their protected status under Queensland legislation such as the *Fisheries Act 1994*. However, they also provide habitat for disease vector mosquitoes such as *Ochlerotatus vigilax* (Skuse). Hydrology is also the driving variable for mosquito populations and is essential in the immature stages.

By modifying hydrology in intertidal wetlands mosquito populations can be managed. Examples abound in salt marshes where open water marsh management (OWMM) is used in the USA (Ferrigno & Jobbins 1968) and runnelling—a minor form of OWMM—is used in parts of Australia (Hulsman *et al.* 1989). Thus, to manage the mosquito population in mangrove systems it is necessary to understand the hydrology of the system, both at a system level and at a micro or mosquito-habitat level, so as to plan and implement appropriate mosquito control.

Lewis III (2005) has provided a comprehensive review of the literature on mangroves and their hydrology and notes that they are "flushed frequently by tidal waters and usually would be inundated on the highest tides" (Lewis III 2005 p. 405). However, the situation is a complex one. As early as 1928, Watson (1928) identified five classes of tidal inundation in mangroves ranging from flooding on every high tide to only occasionally flooded. In his 1966 paper, Macnae (1966) argued Watson's classification was limited pointing out that it was "clumsy" and that as Watson's classification "was devised for the ever-wet Malaya, [it] clearly does not apply so well to regions with drier climates" (Macnae 1966 p. 91).

Rather than using flooding status, Macnae's alternative was to use dominant tree species on which to base his mangrove zonation schema. In so doing he combined topographic position (from landward fringe through to seaward fringe) and dominant mangrove species (ranging from landward *Avicennia, Ceriops* and *Bruguieria* through to seaward *Rhizophora*—sometimes *Avicennia*). However, Macnae's schema was specific to tropical northeastern Australia, an area where mangroves "reach their most characteristic development" (Macnae 1966 p. 67) and is difficult to apply in areas where mangroves have a less characteristic development. For example, in southeastern Australia mangrove forests may comprise few species and at their southern extent just a single species (*Avicennia marina*).

Yet considerable variations in mangrove forest form, structure and hydrology exist that neither of these early approaches capture, particularly when considering mangroves at higher latitudes. A more generic approach is needed to capture the general form of mangrove zonation across the full extent of intertidal possibilities. To some degree this was resolved by Lugo and Snedaker (1974), when they proposed a schema with six mangrove forest forms for south Florida mangrove communities based on local hydrology, particularly patterns of tide and surface drainage. These have been summarised in Table 4-1. By defining mangrove zonation in terms of forest structure and hydrology, Lugo and Snedaker combine aspects of both Watson and Macnae by incorporating topographic form, position and inundation characteristic into a broader landscape. By delineating basin forest and hammock basin forest as hydrologically distinct classes of mangrove forest, Lugo and Snedaker take into account a range of hydrologic conditions not easily considered otherwise.

Table 4-1. Summary of mangrove forest forms identified by Lugo and Snedaker (1974)

Mangrove forest type	Hydrologic characteristics
Fringing forest	Occur along fringes of protected shorelines; most developed when elevations are higher than mean high tide; low tide flow velocities; large accumulation of debris
Riverine forest	Tall floodplain forests along drainage lines; usually separated from channel by a shallow berm; flushed daily by tide; low surface water velocities; often adjacent to and inland of fringing forests
Overwash forest	Forest occupying land masses (e.g. small islands) in shallow estuary water; their position obstructs tidal flow resulting in over-washing at high tide; little debris persists as it is usually washed away by high tide and not returned by the ebb tide
Basin forest	Inland areas along drainage depressions that channel runoff towards the coast; in more coastal locations may be inundated by daily tides; tidal influence decreases with distance inland
Basin forest— hammock	Exists along topographically flat coastal fringe; similar to basin forest type but occurs on ground that is slightly elevated (~5–10cm) with respect to surrounding areas
Dwarf forest	Exist along flat coastal fringe; all individual plants are stunted and relatively old; no obvious external source of nutrients

Nevertheless there is little field data regarding tidal flooding patterns and there have been few attempts to quantify this (Lewis III, 2005). Lewis was of the view that neither ecologists nor engineers fully understand mangrove hydrology, particularly the role of tidal flooding, and hence attempts at management may be inappropriate. Lewis asserted that flooding depth, duration and frequency are critical factors in a mangrove ecosystem and this was supported by Field (1998), who noted that tide height is critical for mangrove survival. However, Perdomo *et al.* (1998) pointed out that mangroves can grow in shallow permanent water and they also noted that parts of their study area were without much connection to water bodies, though rainfall was important.

Twilley and Chen (1998), researching mangrove forests in south west Florida, recognised a need to identify how hydrology affects ecological processes in coastal wetlands. Basin forested wetlands cover large areas of south Florida but they are not well understood because of their complexity. There are few quantitative water budget studies and this limits understanding of the role of hydrology. Twilley and Chen (1998) found that tides did not flood the forest until a threshold level was reached and this was related to local berm formations.

They concluded:

> "Understanding the relative influence of tides and precipitation on the hydrology of the upper intertidal zone is important to understanding the ecological patterns of coastal wetlands" (Twilley & Chen 1998 p. 320).

Surface flow across mangroves generally occurs as a result of rainfall events and during flood tide periods when the mangrove forest is under water. With additional rain or tidal water, more and more depressions are connected and a network of flow paths is eventually formed. Surface roughness is an important factor determining the surface flow and this was modelled by Darboux *et al.* (2002), though not in an intertidal context. However, it is important in mangrove systems and Furukawa *et al.* (1997) showed that the density of above-ground roots has an effect on the water movement, creating a large amount of friction.

Perdomo *et al.* (1998) noted that rainfall is important in the tropical system in Colombia which they studied, and that tidal influence was less important. Cahoon and Lynch (1997) also noted the importance of rainfall in a southwestern Florida mangrove system. They observed a variable hydroperiod based on frequency, duration and depth of flooding. The basin forest, behind a natural berm, had less than one-tenth as many flood events when compared with fringing and island mangrove forest types. The basin type was flooded only by the highest tides but was flooded for the longest period (up to 10.6 weeks) with rainfall as the primary control on mean water depth. Other factors that influenced hydrology included the shrinking and swelling of sediments and the role of subsurface water in affecting local elevation.

The Coombabah Lake area is one that has experienced considerable urban development over the last decade (Lee *et al.* 2006). As it is also a major mosquito larval habitat, the human health issue has become urgent. The need to focus control within the system is important and this research is the first stage of identifying microhabitats in which mosquito breeding is likely to occur. These are areas that hold water and may be disconnected from the tidal source for at least some of the time. Previous research at the study site using thermal scanning and synthetic aperture radar has indicated that the pattern of water under the mangrove canopy is complex (Knight *et al.* 2000; Dale *et al.* 2005; Dale *et al.* in press).

Approach and methods

The study area

The study area is a system of mangrove forests associated with Coombabah Lake. The forest is comprised mainly of the grey mangrove (*Avicennia marina* (Forsk.) forming dense stands in low-lying areas connected to the lake. A mix of mangrove species including *Avicennia, Ceriops, Bruguieria* and *Rhizophora* are also found in fringes immediately adjacent to the lake edge. The lake edge-mangrove interface is around 19.5 km in length with mangroves extending inland for a distance ranging between about 5 m and 700 m, and commonly extending to 400 m inland. Vegetation communities surrounding the lake (based on Dowling & Stephens 1998) and extending from lower to higher elevations include: mangroves (*Avicennia marina* dominant, 339 ha); salt marsh (118 ha); swamp oak (*Casuarina* dominant, 79 ha); paperbark (*Melaleuca* dominant, 168 ha); and then eucalypt forest (238 ha) occupy the higher ground.

Mangrove forest forms

An initial assessment of mangrove forest form, based on Lugo and Snedaker (1974), was undertaken to identify different substrate structural forms and to guide tide gauge deployment. Details of the forest type associated with each tide gauge site are recorded in Table 4-2 as part of the site hydrologic condition description. In all, three of the broad mangrove forest forms identified in Table 4-1 were found to be present within the Coombabah Lake wetlands.

Overwash forests were found on two small islands located within the lake just north of Site 18 (see Figure 4-1). Fringing forests were found extending from the lake edge around Sites 15 and 17 and in a thin strip around the lake edge between the northeastern and northwestern forest sections (Figure 4-1).

Basin forest forms were found in most areas on the landward side of fringing mangroves usually an obvious berm was found between the fringing forest and the beginning of the basin forest. Two forms of basin forest were identified: a basin with a relatively deep standing water body (up to 50 cm deep), and a shallow basin where standing water was much shallower (between about 10 and 20 cm deep). The substrate structures of the two basin forms were markedly different, with the basin substrate structure comprising mostly water and mangrove pneumatophores (northeastern, eastern and central forest sections) and the shallow basin form substrate was composed primarily of water, exposed mud sediment and pneumatophores in about equal proportions (northwestern, southern and southwestern forest sections). The berm associated with the shallow basin form was much less pronounced than for the basin form. The central mangrove forest ranged between a standard (deeper) basin in the northern part around Site 9, becoming progressively shallower to the south and eventually ending as a relatively wide strip of fringing forest around Site 17 (see Figure 4-1).

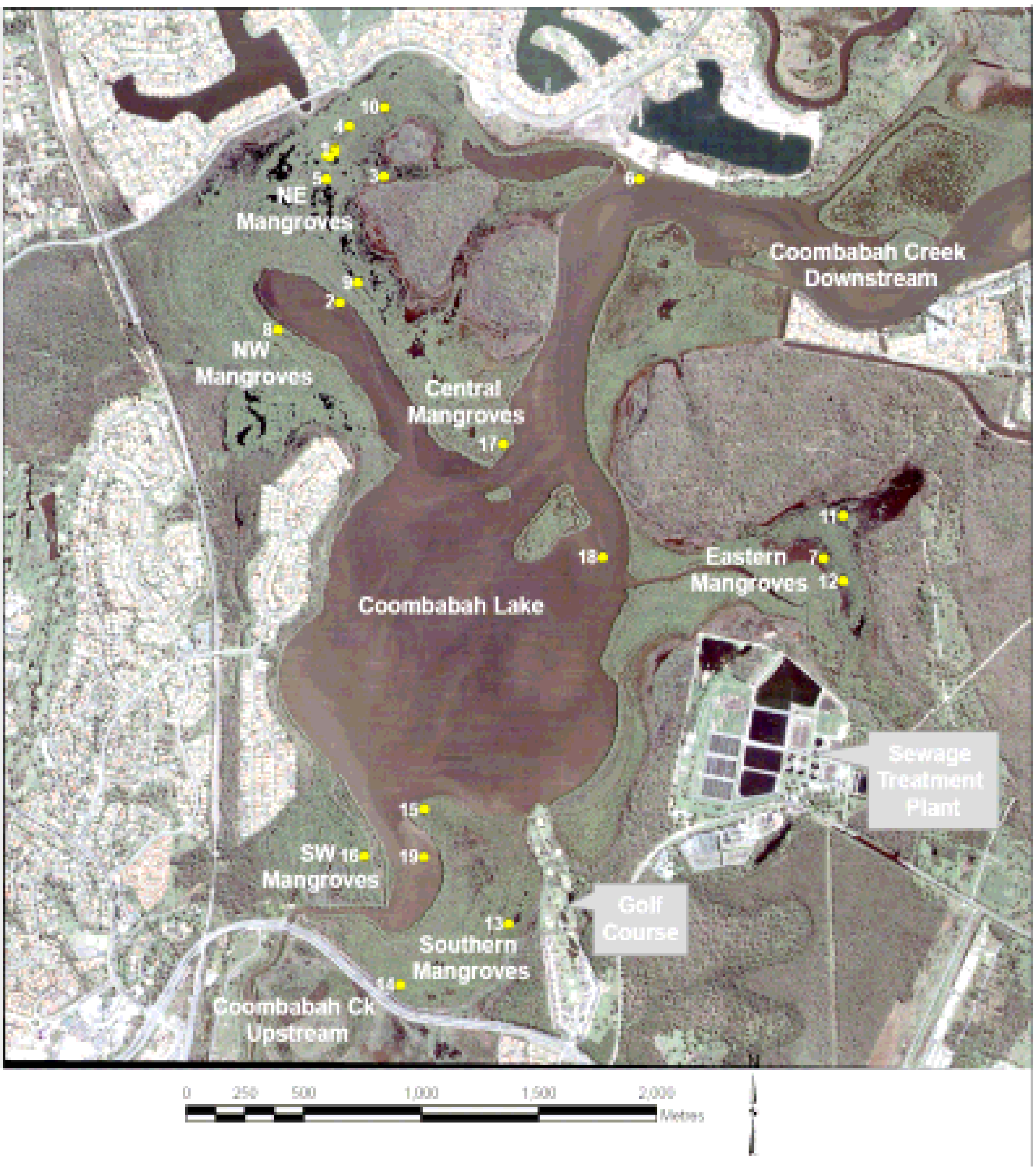

Figure 4-1. Coombabah Lake, showing location of tide gauges and mangrove areas

Tide gauge deployment

Because of the limited number of tide gauges available for any particular monitoring deployment, deployment of gauges was prioritised using a combination of predicted tide height and prior monitoring results. Assessment of prior monitoring included analysis of antecedent inundation and tidal flooding hydroperiod data (frequency, duration and depth of flooding). In this way an incremental strategy evolved whereby gaps identified were given top priority and an increased coverage of lake-edge mangroves was achieved secondarily.

A total of 21 deployments occurred at 18 locations in the study area (see Figure 4-1). In most cases tide gauges were deployed in pools located within areas of mangroves so that an indication of the persistence of the pool and the change in water depth due to flooding could be estimated. One site (Site 1, pool 3) was sampled on four occasions while all others were sampled once. Tide gauge deployments occurred during six sampling periods between July 2002 and November 2005. In most instances deployment was timed to coincide with spring tide cycles. Details of each deployment are reported in Table 4-2, including a brief description of the hydrologic setting and estimate of distance from the nearest tidal source.

Data collection

Official tidal data

Official tide data were acquired for the Gold Coast tidal station (Marine Operations Base, Southport tidal station #100035) from the Tidal Unit, Maritime Safety Queensland (MSQ), and included tide predictions and tide observations corresponding to each tide gauge deployment in the study. The Tidal Unit, MSQ, also provided 19 years of observation data for high tides above 1.3 m and high and low tidal exceedence data. The 19 years of observations extended for 6940 days from 1 January 1986 through until 31 December 2004. A total of 12 520 high tides were recorded, with no readings on 534 days (John Broadbent, MSQ, pers. comm.).

Tidal observation data for the period 5–19 June 1998 from a tidal study conducted in the southern section of Coombabah Lake at a location known as ‘the crab farm’ (tidal station #001027A) were also provided by the Tidal Unit, MSQ. An estimate of variability in tidal predictions for the Gold Coast tidal station, based on 2004 predictions and observations, were provided by the National Tidal Centre, Bureau of Meteorology (<www.bom.gov.au/oceanography/tides/>).

Table 4-2. Tide gauge deployment details including hydrologic condition, dates of deployment and estimated distance from lake-edge tidal source

Site ID	Site hydrologic condition	Deployment dates (No. of days)	Distance from lake edge
Northeastern forest			
1A	Permanent pool ~0.3 ha; average water depth 41 cm (± 5 cm) when not flooded; in basin forest	18/07–01/09/2002 (45)	550 m
1B		02–15/09/2002 (13)	
1C		14/02–18/03/2003 (32)	
1D		07–17/01/2005 (10)	
3	Basin forest—permanent water ~20 cm	02–15/09/2002 (13)	370 m
4	Basin forest—permanent water ~20 cm	02–15/09/2002 (13)	530 m
5	Permanent pool ~1.5 ha; average water depth 42 cm (± 6 cm) when not flooded; in basin forest	02–15/09/2002 (13)	420 m
10	Basin forest—permanent water; initial depth ~22 cm	07–17/01/2005 (10)	370 m
Eastern forest			
7	Permanent pool ~0.1 ha; average water depth 40 cm (± 5 cm) when not flooded; in shallow basin forest	14–24/02/2003 (10)	230 m
11	Shallow basin forest—permanent water; initial depth ~18 cm	04–14/02/2005 (10)	440 m
12	Shallow basin forest—persistent water; initial depth ~14 cm	04–14/02/2005 (10)	190 m
Northwestern forest			
8	Small ephemeral pool (~10 m^2) in fringing forest; initial water depth 10 cm	07–17/01/2005 (10)	40 m
Central forest			
9	Basin forest—permanent channel; 30 cm deep, 1.5 m wide, ~30 m long	07–17/01/2005 (10)	100 m
17	Fringing forest—exposed at time of deployment	31/10–10/11/2005 (10)	30 m
Southern forest			
13	Small ephemeral pool (~7 m^2) in shallow basin forest; initial water depth 8 cm; near golf course	04–14/02/2005 (10)	330 m
14	Small ephemeral pool (~5 m^2) in shallow basin forest; initial water depth 5 cm; near Gold Coast highway	04–14/02/2005 (10)	240 m
15	Small ephemeral pool (~6 m^2) in fringing forest; initial water depth 4 cm	31/10–10/11/2005 (10)	40 m
Southwestern forest			
16	Permanent pool (~0.1 ha) in shallow basin forest; initial water depth 40 cm	31/10–10/11/2005 (10)	100 m
Waterway			
2	Northwestern arm of lake—exposed mudflat at low tide	18/07–01/09/2002 (45)	-20 m
6	Downstream creek—not exposed at low tide	02–15/09/2002 (13)	-15 m
18	In lake—not exposed at low tide	31/10–10/11/2005 (10)	-130 m
19	Crab farm—southern Coombabah Lake (MSQ data)	05–19/6/1998 (14)	Unknown

Meteorology data

Air pressure and rainfall data were obtained from the Queensland Bureau of Meteorology, Brisbane, for the Gold Coast Seaway Weather Station for each tide gauge deployment. Half-hourly air pressure data, corrected to mean sea level, were used to correct logger pressure readings as described below.

Water depth

Pressure sensors were used to record water depth during the study. Two brands were deployed: Greenspan (PS310 and CTD350) and Ocean Sensor Systems (wave gauge OSSI-010-003B). The Greenspan pressure gauges were set to acquire five readings at two-second intervals every ten minutes, recording the average of the five readings for each ten-minute period. The Ocean Sensor Systems wave gauges were set to acquire readings at two hertz for one minute every ten minutes—the average of the maximum ten readings were used as the reading for each ten-minute interval—and two hertz was the lowest frequency setting for these gauges.

Pressure sensor processing to *in situ* water depth

Greenspan sensors

Raw pressure data were recorded as water depth by the sensor. The raw depth data were corrected for ambient atmospheric pressure as recorded by the Bureau of Meteorology at the Gold Coast Seaway Weather Station, located 10 km southeast of the study site. Raw pressure data were corrected to local depth using Equation 4.1 (final depth adjustment) and Equation 4.2 (air pressure depth correction) below:

$$D_{final(i)} = D_{adj(i)} - (D_{adj(0)} - D_{isdd}) \qquad \textbf{(Eq. 4.1)}$$

where:

$D_{final(i)}$ = Logger recorded depth corrected for air pressure at time i, and for initial sensor depth

D_{isdd} = Initial sensor deployment depth (time i=0)

$D_{adj(0)}$ = Initial logger recorded depth adjusted for air pressure (time i=0) calculated using Equation 4.2

$D_{adj(i)}$ = Logger recorded depth adjusted for air pressure at time i, calculated using Equation 4.2.

$$D_{adj(i)} = D_{raw(i)} - ((AP_{amb(i)} - AP_{ltaap}) \times cf) \qquad \textbf{(Eq. 4.2)}$$

where:

$D_{raw(i)}$ = Raw logger recorded depth at time i

AP_{ltaap} = Local long-term average air pressure (1013.3 Hpa)

$AP_{amb(i)}$ = Ambient air pressure at time i

cf = Air pressure to depth conversion factor (0.0102 m).

Ocean Sensor gauges

For the Ocean Sensor wave gauges, the raw data was recorded as a pressure reading in bars and for these gauges the logger data were converted to a raw water depth value ($D_{raw(i)}$) using Equation 4.3 prior to processing as above.

$$D_{raw(i)} = P_{raw(i)} \times AP_{ltaap} \times cf) \qquad \textbf{(Eq. 4.3)}$$

where: $P_{raw(i)}$ = Logger recorded pressure (bars) at time i;
other terms as above.

Data analysis

Description of standard port high tide dynamics

The Gold Coast tidal station semidiurnal tidal plane data were used to base relationships between tides at the Gold Coast standard port and tides in the study site at Coombabah Lake. The tidal plane was defined according to definitions published in Anon (2005). Long-term high tide observation data were used to derive relationships between tide heights based on three tidal dynamics: number of tides, the percent exceedence of high tides and the number of high tide events.

Predicting number of high tides above trigger heights

The Gold Coast tidal station long-term high tide observation data between 1.3 m and 2 m (in 50 mm increments) were used to model the frequency of high tides exceeding a range of tide height increments. These data were plotted and a second-order polynomial was fitted and used to estimate the number of high tides exceeding trigger heights observed in the study.

Predicting high tide percent exceedence above trigger heights

High tide exceedence data between 1.3 m and 2 m (in 50 mm increments) were calculated from the long-term Gold Coast tidal station high tide dataset. These data were plotted against corresponding high tide heights and fitted with a second-order polynomial. The polynomial was used to estimate the high tide percent exceedence for trigger heights observed in the study.

Predicting number of high tide events interspaced by more than 5 days

The long-term high tide observation database was used to extract estimates of the number of discrete tidal flooding events as defined by counting as a single event all high tides exceeding a designated trigger height that occurred within 5 days of an adjacent high tide as in Equation 4.4.

$$\begin{aligned}&If\ \ Date_{Tide(i)} \rangle (Date_{Tide(i-1)} + 5days), \quad then\, Event_{(i)} = 1,\\&else\, Event_{(i)} = 0\end{aligned} \qquad \textbf{(Eq. 4.4)}$$

where:

$Date_{Tide(i)}$ and $Date_{Tide(i-1)}$ are the dates of consecutive high tides that exceed a designated trigger height

Event is a flag indicating high tides that are preceded by more than 5 days of lower than trigger height high tides.

The numbers of tidal flooding events for high tides between 1.3 m and 2 m (in 50 mm increments) were obtained by summing across the 'event' record for each height interval. These data were plotted and a third-order polynomial was fitted from which the numbers of flooding events for trigger heights observed during the study were estimated.

High tide relationships between the study site and standard port

Unimpeded tidal observations (observations from gauges located where full tidal expression occurred) from three spring tide surveys (June 1998 – Site 19; September 2002 – Site 6; and November 2005 – Site 18) were used to identify high tide timelag and amplitude attenuation relationships between Coombabah Lake the Gold Coast tidal station.

To estimate the timelag, times of high tides (at a resolution of 10 minutes) from the three tide surveys (Sites 6, 18 and 19) were compared with corresponding high tides from the Gold Coast tidal station. A Gold coast high tide minimum of 1.4 m (relative to lowest astronomical tide datum) was used as a cut-off for inclusion in the temporal analysis because this was the height of mean high water spring tides at the Gold Coast tidal station and was lower than any of the trigger heights observed in the study. The cut-off was used to remove the possibility of confounding caused by differences in tide–time relationships at other tide stages (such as during neap tides).

To estimate high tide amplitude differences between Coombabah Lake and the Gold Coast tidal station, high tides for the three full amplitude lake sites (Sites 6, 18 and 19) were plotted against corresponding high tide observations for the Gold Coast tidal station. A linear function was fitted to each of the three datasets from which the slope (first derivative) was used as an estimate of the attenuation in tide height amplitude at Coombabah Lake (John Broadbent, MSQ, pers. comm.).

Lake-edge tidal observations

In order to identify differences in tidal pattern across the different mangrove forest sections at Coombabah Lake, hydroperiod data were compiled from individual tidal traces. For each tide gauge deployment, water heights before, during and after tidal flooding were identified from the tidal trace as shown conceptually in Figure 4-2. The conceptual model illustrates the relationship between water heights and hydroperiod data. In the model, horizontal lines are positioned to represent relative water height relationships in the vertical direction and temporal relationships are represented by position and length in the horizontal direction. Thus, for example, the minimum recorded water height (MRH) was always lower than or equal to the minimum water height prior to the first flooding tide (HPFT) and thus the position of the HPFT line is temporally overlapping the MRH line and sloping upwards from the MRH line. In all traces the MHR occurred prior to the first flooding tide, hence its position at the far left of the model.

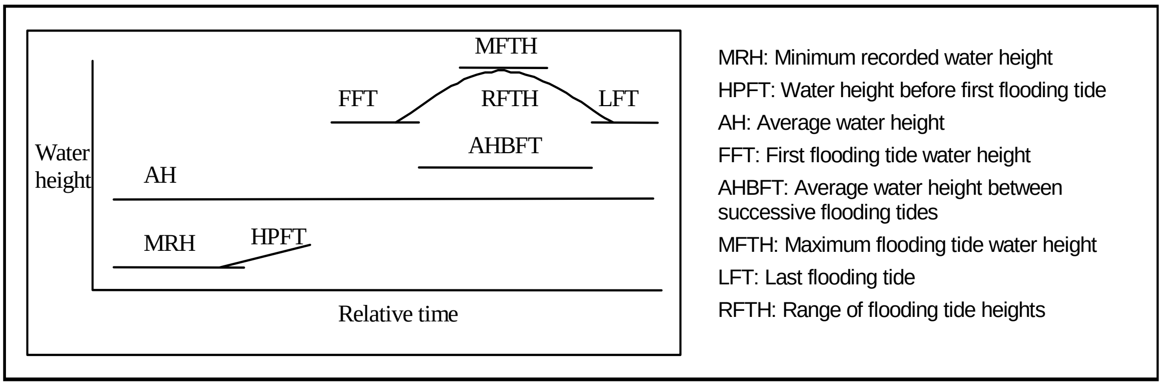

Figure 4-2. Conceptual relationship between water heights and hydroperiod data extracted from each tide gauge trace

The horizontal lines are positioned to represent relative water height relationships in the vertical direction and temporal relationships are represented by position and length in the horizontal direction.

The term trigger height is used to describe the height of the lowest high tide that floods an area of mangrove forest. High tide observations for the Gold Coast tidal station for each trigger height for the period January 1986 to December 2004 were used to identify high tide percent exceedence for trigger height tides. The number of flooding cycles was also derived from the long-term observations data by counting as unique each group of high tides exceeding the trigger height and temporally separate from the preceding trigger height high tide by a period of more than 5 days. This provided an estimate of the frequency of flooding events where groups of consecutive tides above a trigger height were considered part of the one flooding event.

Results and discussion

The Gold Coast tidal station—the ‘standard port’

Details of the semidiurnal tidal plane for the Gold Coast tidal station are shown in Table 4-3. Only high tides greater than the mean high water springs (MHWS) (1.41m) caused flooding into mangrove areas. It is for this reason that the analysis considers tidal heights above 1.3 m.

Table 4- 3. Semidiurnal tidal plane for the Gold Coast standard port (height datum set at lowest astronomical tide (Source: Anon 2005)

	HAT*	MHWS*	MHWN*	MSL*	AHD*	MLWS*	MLWN*	LAT*
Height (m)	1.89 m	1.41 m	1.15 m	0.85	0.76 m	0.49 m	0.23	0.0 m

*HAT = highest astronomical tide
MSL = mean sea level
AHD = Australian height datum
LAT = lowest astronomical tide

MHWS = mean high water springs
MHWN = mean high water neaps
MLWS = mean low water springs
MLWN = mean low water neaps

The annual (pa) average frequency of high tides by height for high tides recorded at the Gold Coast tidal station for the period from 1986 to 2004 is shown in Figure 4-3. A second-order polynomial line of best fit was fitted to the data between 1.3 and 1.9 m. The annual frequencies range between 301 pa (n = 5730, range 189–394, SE 0.77) for tides 1.3 m and greater to 3.2 pa (n= 61, range 1–14, SE 0.46) for tides 1.9 m and greater. Twenty-nine of the 61 tides equal to or greater than 1.9 m occurred in a three-year period from 1999 (5, 14 and 10 tides per year respectively).

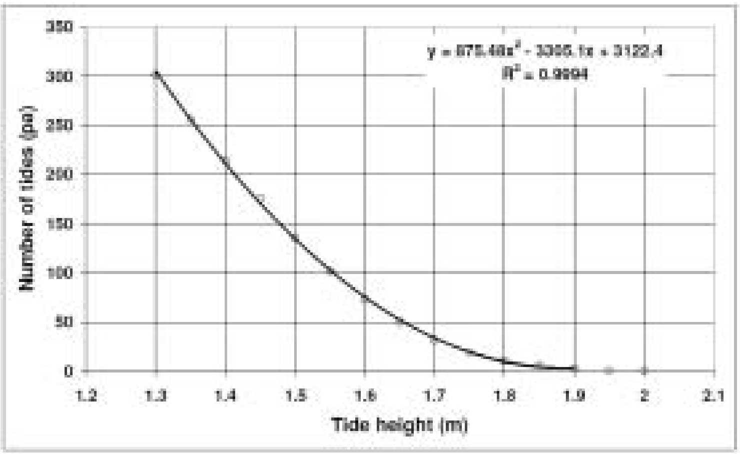

Figure 4-3. Yearly frequency of tide heights for tide heights between 1.3 and 1.9 m based on long-term Gold Coast tidal station observation data

High tide percent exceedence data for Gold Coast tidal station high tides for the period from January 1986 to December 2004 are plotted in Figure 4-4. A second-order polynomial has been fitted to the data. Using the equation shown on Figure 4-4, an estimated 31% of high tides exceed the mean high water springs (MHWS) height of 1.41 m and only 0.45% of high tides exceed the highest astronomical tide (HAT) height of 1.89 m. Given the semidiurnal nature of tides in southeast Queensland the number of tides around 1.3 m includes a number of lower high water tides each day as well as the higher high water.

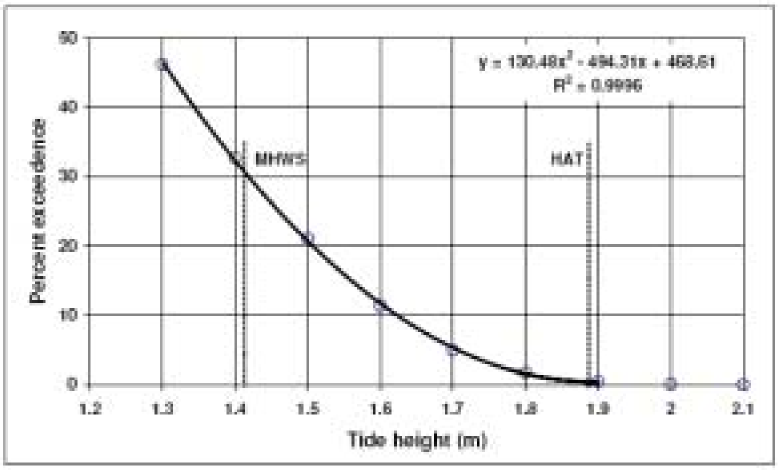

Figure 4-4. Plot of high tide percent exceedence for spring tide heights between 1.3 and 2.1 m based on long-term Gold Coast tidal station observation data

Note that percent exceedence is based on those tides that exceed the specified tide height.

The number of flooding events, that is, sequences of flooding tides with up to 5 days separating flooding tides based on the long-term Gold Coast tidal station observation data set has been plotted on Figure 4-5. A third-order polynomial has been fitted to the data for tide heights between 1.3 and 1.9 m. By solving the first derivative of the equation on Figure 4-5, an estimate of the peak number of flooding periods of 18.5 occurring at a tide height of 1.473 m was calculated. This compares with a peak of 19.2 events occurring at a tide height of 1.5 m when derived from the long-term Gold Coast high tide observation data set.

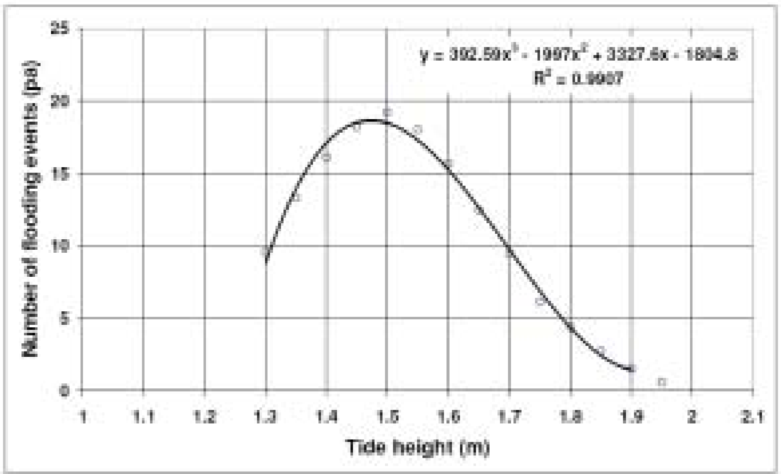

Figure 4-5. Plot of number of flooding tide cycles for tide heights between 1.3 and 1.9 m based on long-term Gold Coast tidal station observation data

Within-lake tidal observations

An average timelag of 2 hours 56 minutes (SD 22 minutes) was observed between high tides at the Gold Coast tidal station and high tides in Coombabah Lake as shown in Table 4-4. As would be expected, the lag for Site 6 (2 hrs 35 min) located slightly downstream was less than the average lag, and the lag for the crab farm (Site 19) was slightly longer than the average being located near the upstream extent of the lake. The average timelag rounded to 3 hours was used to synchronise tidal observations in the current study.

Table 4- 4. Timelags for Coombabah Lake high tides greater than Gold Coast mean high water springs

	2002 Downstream creek (Site 6)	2005 Lake centre (Site 18)	1998 Crab farm (Site 19)	Weighted average
Average lag	2 hrs 35 min	3 hrs	3 hrs 10 min	2 hrs 56 min
Lag standard deviation	18 min	23 min	11 min	23 min
Number of tides	11	8	13	32
Relative position	Nearest ocean	→	Furthest from ocean	

The average attenuation of high tides in Coombabah Lake at 0.82 is the average of the slopes of the three linear equations shown in Figure 4-6. This compares with Maritime Safety Queensland's published attenuation of 0.84 for the Saltwater Creek–Coomera River confluence around 2 km downstream. The slope was used to estimate attenuation in preference to absolute differences because of the difficulty of accurately calibrating recorded heights to the Australian height datum (AHD).

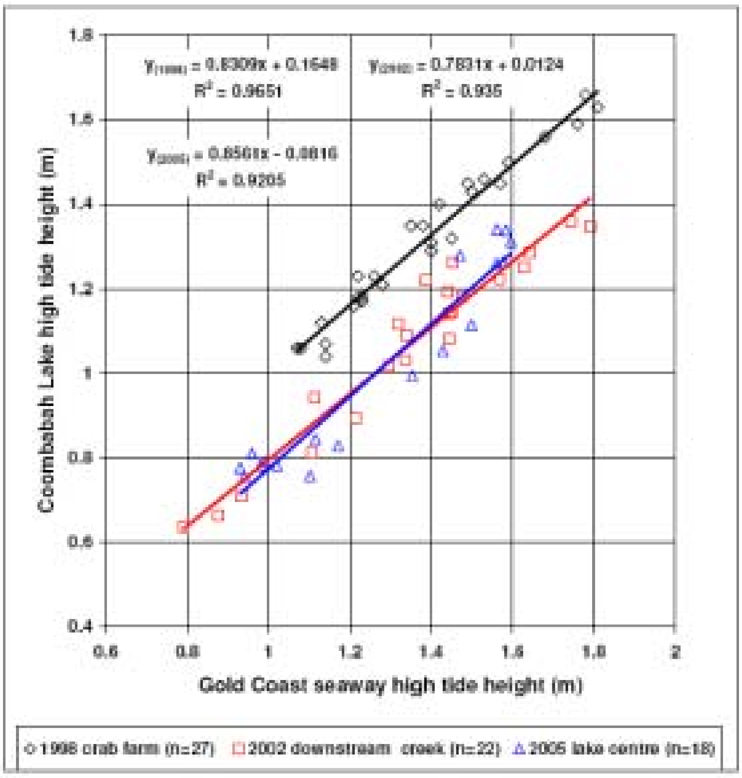

Figure 4-6. Plot of high tide heights for the Gold Coast Seaway high tides against corresponding high tide heights for three sites at Lake Coombabah

A linear line of best fit has been fitted to each site's data with equations shown on the figure. Note that the year of each survey has been subscripted to each dependent variable where Site 6 is Y_{2002}, Site 18 is Y_{2005} and Site 19 is Y_{1998}).

Lake-edge tidal observations

Using the conceptual model set out in Figure 4-2, hydrologic data were extracted from tidal traces and collated in Table 4-5. In addition, corresponding data for tidal parameters related to the local standard port (Gold Coast tidal station) are presented in Table 4-5. The range of Gold Coast tide heights for which tidal flooding into mangrove areas occurred was found to be large, with considerable differences between mangrove sections. A Gold Coast tide height of 1.85 m was required before the northeastern section (Site 1C) was tidally flooded (see lowest flooding tide height, Table 4-5). This compares with the central fringing mangroves (Site 17) being flooded by a much lower tide, with a Gold Coast tide height of 1.5 m.

The relative frequency of such tides is indicated by the percent exceedence data (Table 4-5) where only 0.85% of tides exceeded the tide height required to flood the northeastern mangroves compared with 21.1% of tides that would be expected to be higher in magnitude than the lowest tide needed to flood the central fringing mangroves. In a number of instances (Sites 1A, 1B, 3, 4, 5 and 16, Table 4-5), the trigger height at which tidal flooding could occur was not reached.

Differences in trigger height have been related to long-term inundation pattern and to mangrove form. A limitation on the use of trigger height as we have defined it is that tidal conditions between the Gold Coast tidal station and Coombabah Lake can vary considerably because of extremes in local conditions such high winds and heavy rainfall, and as such these need to be considered. For this reason trigger heights should be thought of as indicators of potential flooding, with some variability inherent in the estimates.

Table 4- 5. Lake-edge mangrove hydrologic characteristics for surveys undertaken during this study, with relationship between characteristics as shown in Figure 4-2

Site ID	Minimum recorded height (MRH) (m)	Height prior to flooding tide (HPFT) (m)	Mean height (AH) (m)	Average height between flooding tides (AHBFT) (m)	Maximum recorded height (MFTH) (m)	Flood range (RFTH) (m)	Duration of flooding	Highest Gold Coast tide (m)	Highest non-flooding tide (HNFT) (m)	Percent exceed-ence HFNT	Lowest flooding tide (LFT) (m)	Percent exceed-ence LFT	Flooding height less non-flooding height (m)
Northeastern mangroves—Basin form behind berm													
1A	0.494	No tidal flood [(a)]	0.470	—	0.517	0.023	—	1.818	1.818	1.30%	—	—	—
1B	0.494	No tidal flood	0.513	—	0.530	0.036	—	1.791	1.791	1.50%	—	—	—
1C	0.360	0.380	0.407	0.410	0.600	0.240	15 hr	1.742	1.850	0.93%	1.856	0.85%	0.006
1D	0.400	0.410	0.440	0.450	0.480	0.080	R-S [(b)]	1.931	1.742	3.44%	1.626	9.36%	-0.116 [(c)]
3	0.260	No tidal flood	0.285	—	0.310	0.049	—	1.791	1.791	1.50%	—	—	—
4	0.061	No tidal flood	0.137	—	0.215	0.154	—	1.791	1.791	1.50%	—	—	—
5	0.500	No tidal flood	0.526	—	0.540	0.040	—	1.791	1.791	1.50%	—	—	—
10	0.185	0.210	0.227	0.259	0. 268	0.083	R-S	1.931	1.850	0.93%	1.856	0.85%	0.006
Eastern mangroves—Basin form behind berm and fringing mangrove foreshore													
7	0.370	0.370	0.420	0.425	0.580	0.210	R-S	1.742	1.564	14.92%	1.682	6.44%	0.118
11	0.171	0.176	0.215	0.220	0.364	0.193	9–10 hr	1.962	1.642	8.29%	1.758	2.91%	0.116
12	0.131	0.135	0.172	0.175	0.328	0.197	10–12 hr	1.962	1.642	8.29%	1.758	2.91%	0.116
Northwestern mangroves—Shallow basin form behind berm and fringing mangrove foreshore													
8	0.073	0.082	0.113	0.130	0.257	0.184	3–4 hr	1.931	1.681	6.05%	1.753	3.07%	0.072
Central mangroves—Basin behind berm and fringing form													
9	0.290	0.296	0.327	0.340	0.493	0.203	5–6 hr	1.931	1.681	6.05%	1.753	3.07%	0.072
17	0	0	0.010	0	0.208	0.208	1–4 hr	1.596	1.430	28.50%	1.500	21.10%	0.070
Southern mangroves—Fringing and shallow basin form behind berm and fringing mangrove foreshore													
13	0.067	0.069	0.114	0.100	0.398	0.331	7–8 hr	1.962	1.449	27.90%	1.590	12.08%	0.141
14	0.046	0.048	0.117	0.085	0.429	0.383	10–12 hr	1.962	1.449	27.90%	1.590	12.08%	0.141
15	0.010	0.030	0.105	0.150	0.219	0.209	4–5 hr	1.596	1.500	21.10%	1.563	14.45%	0.063
Southwestern mangroves—Shallow basin form behind saltmarsh strip and fringing mangrove foreshore													
16	0.408	No tidal flood	0.421	—	0.434	0.026	—	1.596	1.596	12.28%	—	—	—

(a) No tidal flood: No tidal influence observed during deployment
(b) R-S: Water receded slowly after tidal flooding (longer than the period before the next flood tide)
(c) Negative difference: Observations were influenced by a significant rainfall event and considerable local flooding, causing a very significant backwater antecedent

In all cases except for the southwestern mangroves (Site 16) another tidal trace from a nearby tide gauge survey was able to fill in the missing data needed to describe tidal flooding characteristics for each mangrove region. A description of tidal flooding in the southwestern section is limited to the highest non-flooding tide height.

Typically the shape of the tidal trace of sites located where there was impeded flow took the form of a saw-tooth shape where there was a relatively steep rising phase and then a much slower falling phase. The time taken for flooding water heights to recede—or duration of flooding—varied considerably with some sites (Sites 8, 15 and 17), receding to a relatively stable level in a few hours. But for other sites (Sites 1C, 11, 12, 13 and 14), between 8 and 15 hours were needed for the floodwater to drop. In extreme cases (such as at Sites 1D, 7 and 10) the floodwater receded so slowly that water levels had not stabilised before the next flooding tide flooded the site. Except for Sites 13 and 17 all mangrove sites exhibited a higher base-water level (average height between flooding tides, Table 4-5) after being tidally flooded.

The timelags for times of high tide peak between that at the Gold Coast tidal station and mangrove areas such as the northeastern section were up to twice as long (up to 6 hours) as the average lag in the lake proper (~3 hours, Table 4-4). The longer lag times are a result of water flow impeded by a substrate dominated by high densities of pneumatophores (1000–3000 per m^2) and poor connectivity (Jon Knight, unpublished data).

A summary of the hydrologic characteristics of each of the mangrove forest sections is presented in Table 4-6. Gold Coast tidal station high tide trigger heights were identified for each section and used to estimate typical flooding scenarios. In some instances (central and southern sections) two scenarios were identified based on differences in trigger height and flooding characteristics. For example, comparing results for the two central section sites (Sites 9 and 17), it was found that one operates as a basin system (Site 9) with a trigger height of 1.753 m and a flood duration of 5–6 hours (Table 4-5) compared with the other (Site 17) behaving as a fringing system with a trigger height of 1.5 m and a flood duration as low as 1 hour (Table 4-5).

A comparison of the expected number of flood events per year (calculated from the equation in Figure 4-5) with the actual number of flood events occurring per year derived from the Gold Coast long-term record of high tide data is included in Table 4-6. This comparison is presented to give an indication of the reliability of estimates derived using the model with those from the long-term record.

Table 4- 6. Summary of lake-edge mangrove tidal flooding for observed trigger height tides

Mangrove forest section	Mangrove form	Site ID	Highest non-flooding tide (m)	Trigger height (m) [(a)]	No. of tides per year [(b)]	Percent exceed-ence [(c)]	Flooding cycles per year [(d)]	Long-term record-based cycles per year [(e)]
Northeastern	Basin	1C, 1D, 10	1.850	1.856	2.7	0.85%	2.2	2.5 (0.3)
Eastern	Basin	7, 11, 12	1.642	1.682	40.4	6.44%	11.0	10.8 (-0.2)
Northwestern	Shallow basin	8	1.681	1.753	18.8	3.07%	6.9	6 (-0.9)
Central	Basin	9	1.642	1.753	18.8	3.07%	6.9	6 (-0.9)
	Fringing	17	1.430	1.500	135.2	21.10%	18.5	19.2 (0.7)
Southern	Shallow basin	13, 14	1.449	1.590	81.2	12.08%	15.8	16.4 (0.6)
	Fringing	15	1.500	1.563	96.0	14.45%	16.9	17.4 (0.5)
Southwestern	Shallow basin	16	1.596	>1.596 [(f)]	<78.1	<12.28%	<15.6	<15.8 (0.2)

(a) Trigger height is lowest observed Gold Coast tidal station height for which flooding was observed at the study site
(b) Number of tides per year predicted using equation in Figure 4-3
(c) Percent exceedence predicted from equation in Figure 4-4
(d) Number of flooding cycles calculated from equation in Figure 4-5
(e) Derived from long-term Gold Coast tidal station high tide observation data
(f) Site 16 experienced no tidal flooding during observation period, indicating that trigger height tide reported is higher than highest tide observed (1.596 m), so predictions of number of tides, percent exceedence and number of cycles all based on a trigger height of 1.596 m (considered conservative estimates as true values expected to be less than those shown)

A schematic representation of the tidal flooding characteristics of each of the mangrove sections studied against the Gold Coast semidiurnal tidal is presented in Figure 4-7, where tidal flooding into the lake-edge mangroves occurs between an elevation of mean high water spring (MHWS 1.41 m) and the highest astronomical tide height (HAT 1.89 m). Results suggest there are at least four (and maybe more) hydrologic patterns evident within the Coombabah Lake mangrove system, as shown by the groupings of mangrove sections with similar trigger heights.

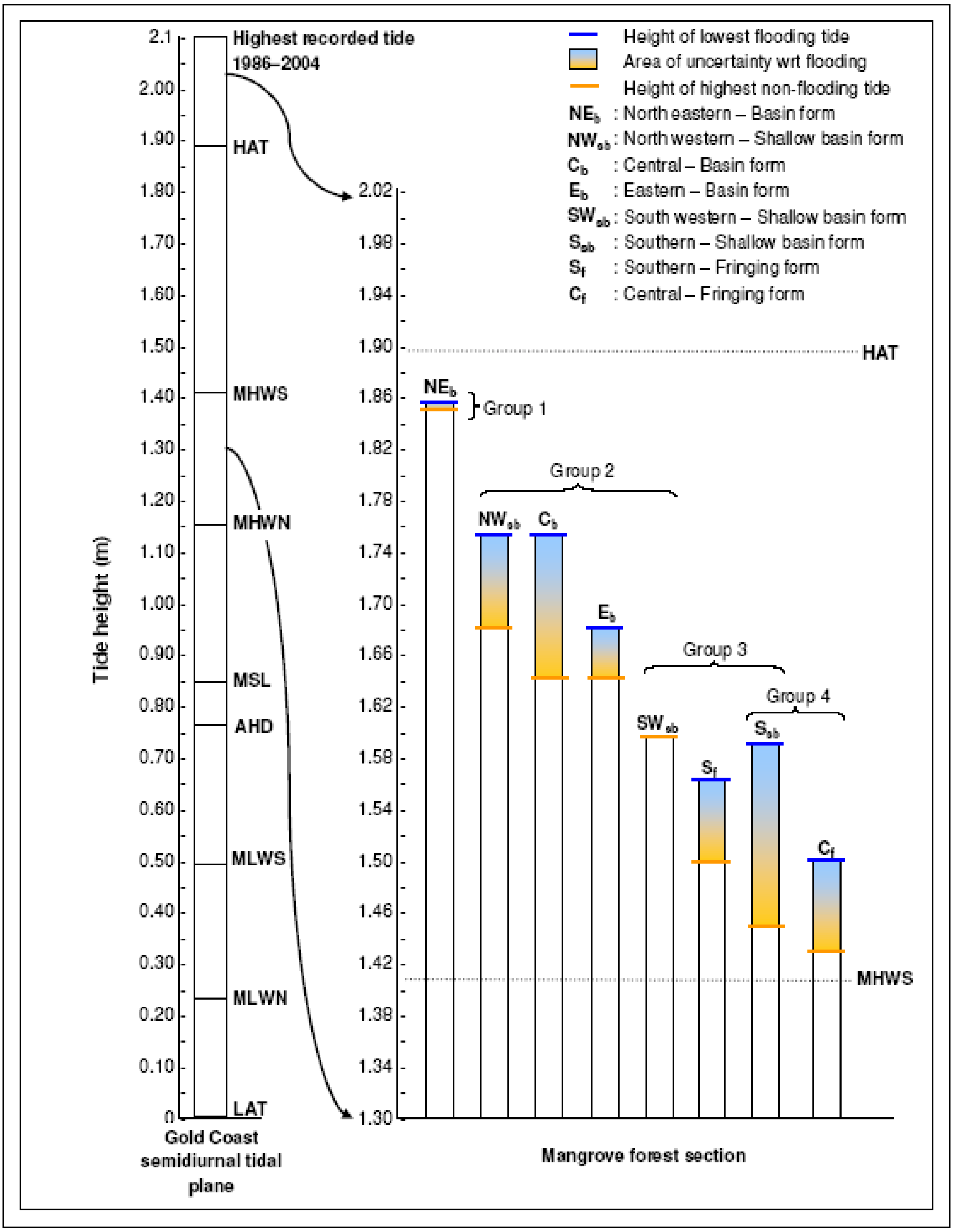

Figure 4-7. Comparison of tidal flooding characteristics of lake-edge mangrove sections in relation to the Gold Coast semidiurnal tidal plane

Group numbers indicate similarities in tidal pattern, with overlaps indicating uncertainty about membership within groups. In most instances an area of uncertainty is shown between the lowest flooding height observed and the highest non-flooding height observed.

The results indicated two main points: first, that the mangrove system at Coombabah Lake is not always flushed by tides; and second, that very distinct hydroperiod characteristics have been observed and that these relate to tidal elevation and substrate form.

Lake-edge mangrove wetlands such as those at the study site are inherently difficult to study because of the difficulty of accurately measuring tidal flow across the wetland surface into or out of the lake. This is particularly so where differences in substrate form as found at Coombabah Lake occur. For example, fringing mangroves are found immediately adjacent to the lake's edge and basin forest mangroves are found on the landward side of the fringing mangroves. In addition, it was found that there are two distinct forms of basin forest based on observed tidal characteristics: those inundated with relatively deep water (around 40–50 cm) and with relatively infrequent exposure to tidally driven flooding; and those with relatively shallow water (around 10–20 cm) with relatively frequent exposure to tidal flooding. The implications for the ecology of these systems have to be considered in light of sea level rise and other possible consequences of climate change. Also, the implications of the variability in mangrove substrate form need to be considered with regard to the effects of urbanisation and wetland management in general.

The implications for mosquitoes are that there appear to be several habitats that seem to be particularly suited to the species found at the Coombabah Lake site. Mosquitoes require water for larval development. Some, such as *Ochlerotatus vigilax*, also have stringent conditions for the egg stage: these are a damp surface on which to lay eggs and then a drying period followed by flooding, such as normally occurs with tidal activity. The species thus thrives in areas where water levels fluctuate.

In the study area, the tidal fluctuations varied considerably and the areas of permanent water were not the ones with most mosquito larval abundance. This can be explained by the lack of water-level changes, in that area water levels rose after rainfall but not after the highest tides. Thus, from a mosquito control perspective, such sites would not be a high priority for larval control.

Conclusion

Highlights of the research include:

- An increased understanding of the complexities of the mangrove forest at Coombabah Lake and the implications of this knowledge for other mangrove systems
- The incremental development of a methodology, as questions arose during the research that can provide an efficient and effective protocol for other assessors of intertidal systems.
- The unveiling of the issue of complex connectivity and its role in ecosystem processes
- The value added by the availability of thermal imagery from Australian Research Council (ARC) funding sources
- The development of research synergies with other Coastal CRC researchers, adding value to the present project and paving the way for future collaborations.

Challenges faced included:

- The difficulty of accessing the mangrove systems
- The four types of forest present at Coombabah Lake making it difficult to move in or carry equipment through the mangrove forest area.

Benefits and outcomes

There are two main types of benefits from this research. One is a practical one that can directly inform management. The other is the methodological innovation that has wide application both within Australia and overseas.

From the practical perspective, the management community will benefit from the far greater understanding of the complex relationships between mangrove microtopography and tidal influences. The study has identified at least four discrete types of system in what has tended to be seen as a homogeneous mangrove forest. This is new information and is relevant to a broader range of mangrove systems, both in Australia and in other parts of the world. The four types will have different management strategies, both to conserve their intrinsic ecological values and to minimise the impacts on nearby human populations from insect, health and nuisance hazards.

In the latter case, by focussing insect pest control on only those parts of the system that actually provide larval mosquito habitat, the costs of insecticide usage can be reduced. For example, preliminary estimates for the study site show a potential saving of around 20% of the cost of aerial larviciding in the mangrove systems. Further validating is needed before this will be achieved, as the risk of failure must be minimised in view of the proximity of the human population and the disease status of the mosquitoes.

The research outcomes' international significance has already been evidenced by Knight and Dale being invited to present the findings to a Florida symposium (The 3rd Arbovirus Surveillance and Mosquito Control Workshop, St Augustine, Florida, 22–24 March 2006), as the approach is a novel one not explored elsewhere.

From the methodological perspective, the development of a survey methodology integrating field and aerial thermal scanning survey to assist in untangling the complex relationships in the mangrove or other intertidal system provides a tool that can be used by stakeholders, other managers and by the wider scientific community to further knowledge about mangrove hydrology and habitat.

Thus the outputs from the research include:

- models of the relationships between tidal characteristics at the local standard port and the specific system (Coombabah Lake mangrove forest)
- a methodology that can be applied to other mangrove and intertidal systems.

As well as being of interest to scientists, both of these are of use to managers of intertidal environments.

Mosquito control activities undertaken by the Gold Coast City Council will greatly benefit from the improved understanding of the hydrology of lake-edge mangroves at Coombabah Lake achieved in this research, by enabling significant refinement to target control operations.

Further development

Further research would be useful to explore the generality of mangrove types and to test the operation of the methodology developed here. Identification of the mangrove system's complex connectivity has already led to a collaborative national proposal (currently under consideration) to expand the research and explore connectivity issues throughout east and southeast coastal Australia.

Chapter 5

Detection of urban influx effects in the lake, using biotic indicators

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Chapter summary

Urbanisation in the coastal zone results in direct wetland destruction, but the indirect impacts on the biota are poorly known. There is a significant amount of knowledge on how urbanisation may cause changes in water quality and nutrient regimes of a coastal wetland. It is unclear, however, how such changes may result in alteration of the abundance and condition of key structural and functional components of the biotic assemblage. Managers of coastal wetlands also need to know to what extent urban development would compromise the ability of coastal wetlands to support beneficial ecosystem services.

In this component of the study, four biotic indicators were used for detecting the impact of urban influx from the local catchment on Coombabah Lake: nitrogen stable isotopes of mangrove (*Avicennia marina*) leaves and of tissues (hepatopancreas and muscle) of the ground-dwelling crab *Australoplax tridentata*; chlorophyll *a* concentration of surface sediments; and body condition of *A. tridentata*. Samples for analyses were collected from three influx locations in the lake, each further comprising sites that were at 0 m and 250 m away from the influx point. Sampling was performed during the dry and wet seasons of 2005–2006 in order to capture differences in the amount of urban influx entering Coombabah Lake due to the difference in rainfall.

The indicators suggest that although Coombabah Lake is generally healthy, there are strong signs that local impacts of urban runoff exist. The values of the short-term indicators (nitrogen stable isotope signatures of crab tissues, which are more responsive to increases in rainfall) at 250 m away from the influx points demonstrated values more reflective of near-normal conditions in the wet period, whereas values significantly more indicative of increased urban discharge were typical of the influx sites. These indicators also demonstrate a worsening of the indicator values during the wet period, when increased rainfall served as a vehicle carrying urban pollutants into the lake. The long-term indicators (mangrove leaf nitrogen stable isotope signature and the body condition of a mangrove crab) are more indicative of time-integrated spatial patterns in local habitat condition. Continual development of the catchment of Coombabah Lake is therefore likely to result in gradual deterioration of this coastal wetland, affecting its long-term capacity for supporting the beneficial ecosystem benefits offered to the region.

Background

Coastal wetland ecosystem services and urbanisation: a case study

Coastal wetlands such as mangroves and salt marshes provide many beneficial ecosystem services such as supporting important commercial fisheries through the provision of food and habitat. The ability of coastal wetlands to provide such services is often compromised by anthropogenic activities that may alter the hydrological regime, sedimentation pattern and water quality. Understanding the impact of urbanisation on coastal wetland structure and function is therefore fundamental to the management of these habitats for sustainable ecosystem services. This component of the study attempted to use selected biotic indicators for detecting the impact of urbanisation on coastal wetland ecosystems through a detailed case study of Coombabah Lake, a known fish habitat area.

The original aim of the study placed an emphasis on assessing the use of the lake as a habitat by fish and waterfowl, and the determination of the trophic base of the system. While these areas were still investigated in this study, the focus of the research shifted towards detecting the impact of urbanisation on ecosystem structure and function by comparing the values of selected biotic indicators around local urban influx points. This approach allows an evaluation of urbanisation as a driver in coastal wetland ecosystem structure and function, benefiting managers in their decisions concerning the planning, creation and management of such habitats in areas with significant anthropogenic pressures.

Coombabah Lake has been the focus of many studies in the last decade, most of which were conducted with relatively limited coverage in time or scope (e.g. Gallagher 2001; Waltham *et al.* 2002; GHD 2003). In particular, GHD (2003) conducted a survey of the major groups of organisms inhabiting Coombabah Lake, providing useful comparative information to this study.

Sources of urban influx to the lake

The operation of a sewage treatment plant (STP) adjacent to the wetland's eastern shore and increasing urbanisation of its catchment have altered the quality and pattern of flow of water into the lake, with likely impacts on the ecosystem health of the lake and its capacity to provide ecosystem services (CRC 2004). Two major creeks flow into the wetland from the upper catchment (Figure 5-1). Coombabah Creek is the larger of the two and probably contributes the greatest volume of urban influx. It enters the southern end of the lake and drains a total subcatchment of 2350 ha, of which about 20% is designated for medium-density residential purposes and 37% is used for rural and park living purposes. Thirty-eight percent of the subcatchment is forested and the remaining 5% is used for commercial and industrial activities (GCCC 2005a). Elevated organic matter content levels of 7.1 to 9.0% and nutrient (phosphorus) concentrations between 310 and 999 $\mu g\ L^{-1}$ have been observed in sediments where Coombabah Creek enters the wetland (Dunn 2005). The second creek ('STP Creek') enters the eastern side of the wetland and drains a predominantly (79%) forested subcatchment of 975 ha, of which roughly 20% is designated for rural and park living residential land uses and 1% for industrial purposes (GCCC 2005a).

The Helensvale STP is located along the eastern shore of the lake and also lies within the STP Creek subcatchment. The secondary treated effluent from the plant is pumped to the Gold Coast Seaway where it is released with the rest of the Gold Coast's treated effluent each evening on the outgoing tide. Effluent would enter the creek or lake only through leaking pipes or as overflow discharge when the capacity of the plant is overstretched at times of high rainfall. However, elevated levels of nitrogen have been observed in the water column near the STP Creek entrance into Coombabah Lake ($0.8 \pm 0.0\ mg\ L^{-1}$) (GCCC 2005b) and elevated organic matter (7.1 to 9.0%) and phosphorus (310 to 999 $\mu g\ L^{-1}$) levels have also been recorded in the sediment in this area of the lake (Dunn 2005). Along the western shore of the lake, two stormwater pipes flow about 5–10 m above the upper tidal limit of the saltmarsh, draining the adjacent Helensvale residential area. This area of the Coombabah catchment is predominantly used for medium-density residential purposes.

Impact on ecosystem health

Previous observations (Dunn 2005; GCCC 2005b) indicate that nutrients are a significant component of urban influx to the lake as a result of land use within the catchment. The current study set out to investigate whether the influx of urban nutrients is having a localised impact on the ecosystem health of Coombabah Lake. This was achieved by comparing the nutrient sources for primary producers and consumers, the biomass of benthic microalgae and the condition index of an indicator consumer sampled from sites close to and far from sources of urban influx. These variables were compared between wet and dry seasons, as a greater amount of urban influx is likely to enter the wetland during the wet season. Data from this study will assist in identifying the dominant sources of urban influx entering Coombabah Lake and assess its condition using benchmark values for wetlands of a similar nature.

Approach and methods

Detecting the impact of urbanisation on coastal wetland ecosystem structure and function present significant methodological challenges. The manipulative experimental approach is usually impractical at the ecosystem level, and most studies are therefore dependent on the use of a rigorously designed sampling program for detecting impact. As long-term studies that incorporate a period of measurement before urbanisation occurs are logistically and financially demanding and are therefore rarely undertaken, detection of impact would in most cases have to rely on spatial comparisons alone (Green 1979).

Spatial comparisons may be made at the across-wetland level—where independent wetlands with differing levels of urbanisation, as measured by some established index or parameter, are compared using some indicators of condition. Quantitative indices are preferred to qualitative classification of wetlands into simple urbanised or non-urbanised states, as the former would allow a correlation between urbanisation intensity and ecosystem condition. Replicated measurements of condition indicators at the whole-wetland level are, however, extremely demanding because of the logistics involved in repeated sampling over large spatial scales.

An alternative assessment strategy utilising the spatial comparison approach is to attempt to detect differences in condition along local gradients of urban influx within individual wetlands. This approach assumes that the level of urban influx is detectably different at different distances away from influx points (i.e. water circulation is not sufficient to distribute influx components evenly across the entire wetland). The value of selected indicators can then be correlated with the assumed influx gradient so as to assess the impact of urbanisation on ecosystem structure and function. Due to limited time and resources, this latter approach has been adopted for the present study, with a detailed study on Coombabah Lake.

Future follow-up research in the region could benefit from the use of several other coastal wetlands with different levels of urbanisation for an across-lake comparative study. Potentially suitable wetlands include Lake Weyba and Lake Cooroibah on the Sunshine Coast, and Terranora Broadwater in far northern New South Wales, which can provide additional comparison locations for Coombabah Lake. These wetlands also present differing levels of urbanisation in their catchment, thus facilitating their use in a gradient design.

Biotic indicators of urbanisation

Three groups of indicators have been selected for assessing the impact of urbanisation both at the lake and local levels at Coombabah Lake: (1) nitrogen stable isotopes ($\delta^{15}N$), both in mangrove leaves and crab tissue; (2) abundance of microphytobenthos (MPB); and (3) the body condition of a common ground-dwelling crab, *Australoplax tridentata*.

Nitrogen stable isotopes—tracers for nutrient sources

Stable isotope analysis is a useful method for identifying sources of nutrients and tracing the flow of organic material in coastal food webs (Fry & Sherr 1984; Macko & Ostrom 1994; Peterson 1999). Stable isotope analysis measures the ratio of heavy and light stable isotopes of certain elements present in a sample of material. For example, ^{15}N:^{14}N (Lajtha & Michener 1994) are commonly used because of their widespread occurrence and importance in biological samples. These ratios are measured in a mass spectrometer, which combusts the samples and converts their elements into gases such as N_2 (Lajtha & Michener 1994). The difference in absolute abundance between the heavy and light isotopes of each element is typically small and subject to variation associated with sample heterogeneity and sample preparation (Hayes 1982).

To account for this variation, the isotopic ratio of each sample is related to a standard (Lajtha & Michener 1994). The primary standard reference material for nitrogen is atmospheric air (Mariotti 1983). The difference between the sample isotopic ratio (R_{sa}) and the standard isotopic ratio (R_{std}) gives the isotopic signature (δ) of the sample material (Equation 5.1) and is measured in units of parts per thousand (‰) (Lajtha & Marshall 1994).

$$\delta = \frac{R_{sa} - R_{std}}{R_{std}} \times 1000 \qquad \textbf{(Eq. 5.1)}$$

The application of stable isotopes to pollution studies has shown great value. Two assumptions, however, must be satisfied in order to use stable isotope abundances as indicators of the origins of materials in the environment:
(1) the primary sources of interest must be isotopically distinct; and (2) the isotopic signature of the source must not change as the material is transported and transformed in the ecosystem or must do so in a predictable manner (Macko & Ostrom 1994; Peterson 1999). While many primary producers share a similar signature for one element, the analysis of additional elements can help to

distinguish them. Theoretically, numerical mixing models can help resolve the relative percentage contributions of different producers to the diet of consumers but the number of sources that may be resolved is limited to the number of elements analysed plus one.

Nutrient enrichment and increasing occurrences of eutrophication in coastal wetlands are attributable to increasing urbanisation in coastal areas (Costanzo *et al.* 2001).The $\delta^{15}N$ of wetland primary producers offers a tracer where the presence of nitrogen derived from urban activities may be determined (Macko & Ostrom 1994; McClelland *et al.* 1997; Costanzo *et al.* 2001). Yamamuro *et al.* (2003) reported that the $\delta^{15}N$ value of seagrass tissues can be used to reflect the level of dissolved inorganic N (DIN) in both the water column and interstitial water. However, because the extent of nitrogen fractionation is difficult to estimate in the natural environment, the use of $\delta^{15}N$ to trace nitrogen sources should only be considered a semi-quantitative or qualitative technique (Macko & Ostrom 1994).

Agricultural fertilisers, animal excrement and treated urban effluent are major sources of nitrate in coastal environments (Macko & Ostrom 1994; Cloern 2001; Costanzo *et al.* 2001). The $\delta^{15}N$ of these nitrogen sources are isotopically distinct from each other, and the $\delta^{15}N$ of treated sewage and groundwater nitrogen generated from human and animal wastes is approximately 10 to 22‰ (McClelland *et al.* 1997) (Macko & Ostrom 1994). Synthetic fertilisers have low $\delta^{15}N$ values ranging from -8 to 7‰ (Macko & Ostrom 1994). The $\delta^{15}N$ signature for groundwater nitrogen influenced only by atmospheric deposition typically has $\delta^{15}N$ values of 2 to 8‰ (McClelland *et al.* 1997).

Assuming that urban influx in wetlands generally contains nitrogen derived from human and animal wastes or from treated sewage, an elevated $\delta^{15}N$ of wetland primary producers and consumers can be used to detect the presence of urban nutrients. This assumption is based on the belief that fertilisers are generally only used in large quantities in agricultural areas. However, if the land in urban areas was oversupplied with fertilisers, the low $\delta^{15}N$ values of the nitrates in fertilisers could mask any elevated $\delta^{15}N$ values of nitrates derived from animal wastes or treated sewage and prevent the detection of urban nutrients in wetlands. Therefore, as a precaution, $\delta^{15}N$ should be used with other indicators of nutrient loading for the detection of nutrient influx in urbanised wetlands.

The stable isotope signature of biological samples is a time-integrated reflection of their nutrient sources, but the temporal responsiveness of the signature to shifts in conditions (i.e. source of nutrients) will be dependent on the turnover rate of the tissues being analysed. Three different sample types were analysed in this study to provide indications of differing time scales. The leaves of the dominant mangrove, *Avicennia marina*, would reflect nutrient sources at the site on the medium (weeks to months) term. Tissues of the hepatopancreas and chelal muscle of the ground-dwelling crab *Australoplax tridentata* are expected to provide, respectively, short (days to weeks) and medium (weeks to months) indication of nutrient sources.

Standing crop biomass (chlorophyll *a* concentration) of microphytobenthos

Due to their small size (≤ 50μm) and rapid turnover rate (hours), microscopic algae and cyanobacteria (microphytobenthos, MPB) respond quickly to environmental fluctuations such as nutrients from urban stormwater runoff (urban influx) (Yang *et al.* 2003). This makes MPB an ideal indicator for short-term temporal fluctuations or pulse effects of urban influx caused by stormwater runoff. Primary productivity is largely controlled by sunlight and the availability of nutrients. Therefore the primary productivity of MPB provides a good indication of the amount of nutrients that are present in a wetland.

A variety of methods are available for the measurement of primary productivity, but these are too time-consuming to be used routinely with enough replication in most surveys of biotic effects of pollutants. An estimation of the standing crop biomass of benthic microalgae is therefore normally used as an indication of primary productivity.

The hepatosomatic condition index in mangrove crabs

The hepatopancreas is an organ in crustaceans that is used for the storage of nutrients and the removal of toxins (Kennish 1997; Connell *et al.* 1999; MacFarlane *et al.* 2000). Therefore the mass of hepatopancreas to body mass expressed as a percentage (hepatosomatic index, HSI) can provide an indication of the nutrient availability of the animal's habitat. Kennish (1997) observed that the peak body condition (as measured by the HSI) of the tropical rocky shore crab *Grapsus albolineatus*, which feeds on filamentous algae, coincided with the annual peak in algal biomass. This author also reported that during the crab's breeding season, a reduction in the HSI was related to an increase in the gonadosomatic index (GSI), demonstrating the transfer of energy from the hepatopancreas to the gonads at this time.

Kyomo (1988) observed a similar pattern in the mangrove crab *Sesarma intermedia.* Due to this transfer of energy between the hepatopancreas and the gonads, both the HSI and the GSI indices varied significantly throughout the year for both species of crab (Kyomo 1988; Kennish 1997). However, due to the higher rate of food consumption by females during the breeding season and greater energy demand for egg production, the HSI was not as variable in male crabs (Kyomo 1988; Kennish 1997). Kyomo (1988) also reported that the GSI seemed to be inversely related to the HSI in female crabs, but the HSI and GSI did not show this relationship in male crabs.

These results demonstrate that the HSI in crab species can be used as an indicator of nutrient availability and thus, the nutrient status of the environment the crabs inhabit. However, this index is also subject to variation associated with the energy demands required for reproduction, more so in female crabs than in males. Therefore, the HSI in male crabs is likely to provide a more accurate indication of nutrient enrichment in wetland ecosystems.

Fish and shore bird assemblages

The fish and shore bird assemblages of Coombabah Lake were surveyed to provide an indication of the level of use of the wetland by these important components of the fauna. Since fully quantitative surveys require extensive spatial and temporal replication, they were beyond the scope of this study and the survey conducted aimed only to append further qualitative information on the assemblages recorded from Coombabah Lake to date (e.g. GHD 2003).

Objective and models

To determine whether urban nutrient influx is having a localised ecological impact at sites where urban influx enters Coombabah Lake and if this impact varies between wet and dry seasons, the following models were tested:

Model 1: The mean $\delta^{15}N$ value of MPB and the mangrove *Avicennia marina* is more enriched in urban influx sites than in sites located 250 m away from urban influx. This effect is greater during the wet season than during the dry season.

Model 2: The mean $\delta^{15}N$ value of the mangrove crab *Australoplax tridentata* is more enriched in urban influx sites than in sites located 250 m away from urban influx. This effect is greater during the wet season than during the dry season.

Model 3: The standing crop biomass (mean chlorophyll *a* concentration) of MPB is greater in urban influx sites than in sites located 250 m away from urban influx. This difference is greater during the wet season than during the dry season.

Model 4: The mean hepatosomatic index (HSI) of the mangrove crab *Australoplax tridentata* is greater in urban influx sites than in sites located 250 m away from urban influx. This effect is greater during the wet season than during the dry season.

Sampling design

Urban influx sites were located within the intertidal zone on both sides of the source of influx at three locations around Coombabah Lake. Reference sites were located 250 m along the shore from where the urban influx entered the lake, on both sides of the influx (Figure 5-1). Each sampling site was approximately 50 m^2 in area and located within the mangrove/mudflat fringe of the intertidal zone. Indicator samples were collected haphazardly from each site during the daytime low tide over a period of 3 days. This sampling was replicated three times during the dry season (September–October 2005) on a two-weekly basis and three times during the wet season (December 2005–January 2006) on a two- weekly basis.

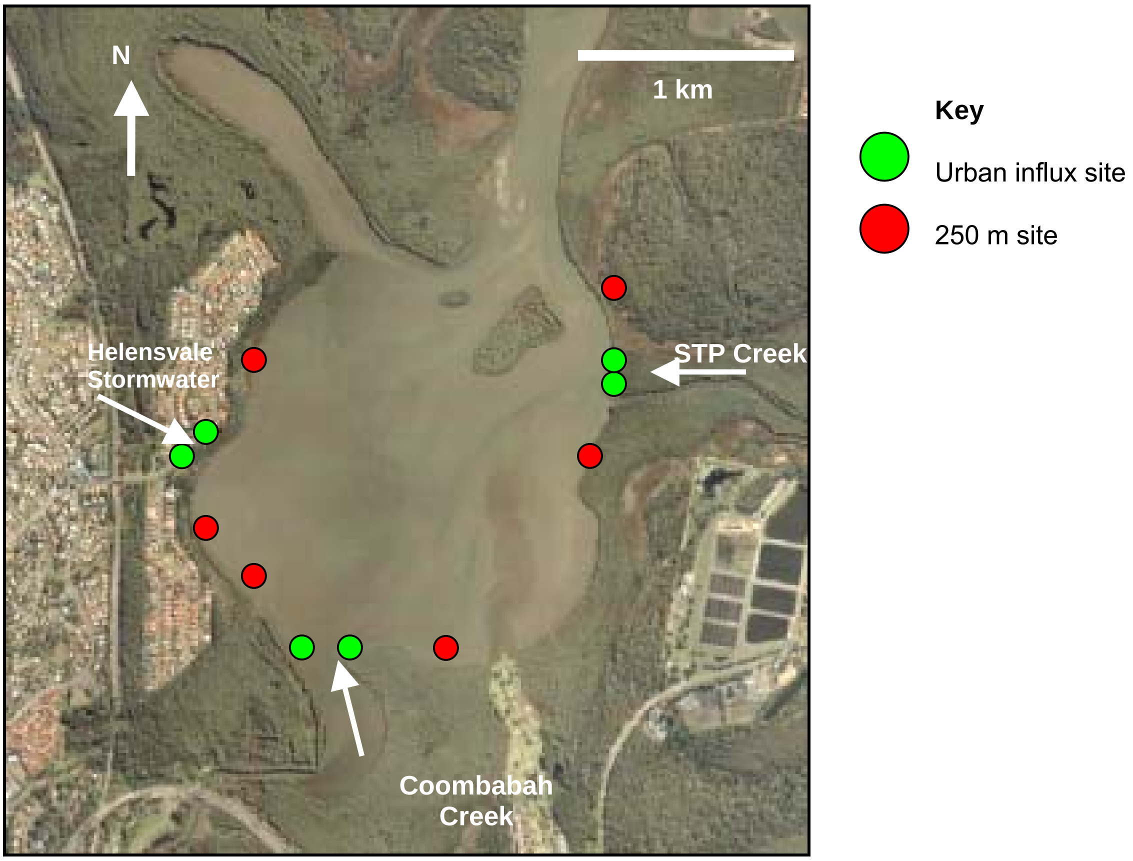

Figure 5-1. Spatial sampling design for detection of the impact of urban influx within Coombabah Lake

Sample collection and processing

Stable isotope analysis

Five leaves were sampled from individual *Avicennia marina* trees in the three locations at the 0 m and 250 m sites. Five *Australoplax tridentata* crabs were hand-collected from within each site. All samples were placed in ice slurry immediately after collection, returned to the laboratory and frozen until analysis.

In the laboratory, all samples were thawed and dried to a constant weight in a desiccating oven at 60°C. Prior to drying, the crab claw tissue and hepatopancreas were removed using stainless steel dissecting instruments and dried separately. After drying, each sample was homogenised using a porcelain mortar and pestle, and a 4 mg subsample of dried homogenised tissue was taken from each mangrove leaf and a 0.6 mg subsample of muscle and hepatopancreas tissue was taken from each crab. The weighed samples were then sealed in tin capsules (Macko & Estep 1984; Lajtha & Michener 1994). Nitrogen isotopic compositions of the samples were measured with a continuous flow GV Isoprime mass spectrometer using ambient air as a primary standard for nitrogen and prawn tissue as a working standard.

Standing crop biomass of MPB (chlorophyll *a* concentration)

Five cores (20 mL) of mud were haphazardly sampled from within each influx and 250 metre site using a modified centrifuge tube as a corer. Samples were placed on ice during transport to the laboratory where they were frozen until analysis.

During analysis, the top 1 cm of mud was removed from each core and transferred to a new centrifuge tube (Yang *et al.* 2003). Fifteen millilitres of 90% aqueous acetone was added and each tube was sealed and shaken vigorously for 1 minute. The tubes were then left to stand over night in the dark at 4°C to ensure effective extraction of chlorophyll from the MPB (Yang *et al.* 2003). After 12 hours, the samples were centrifuged at approximately 4400 rpm for 5 minutes and their absorbance was measured at 750 and 665 nm using a Shimadzu UV-1601, UV-visible spectrophotometer. Measurement of the samples' absorbance at 750 nm accounted for any turbidity and was subtracted from the absorbance values at 665 nm (Lorenzen 1967). The samples were then acidified with 2 drops of 1 M HCl and the absorbances measured again at 750 and 665 nm. These two measurements allowed a correction to be made for absorption by phaeopigments so that active chlorophyll *a* could be distinguished from the chlorophyll in any detrital material in the samples (Lorenzen 1967). Concentrations of chlorophyll *a* (Chl *a*) and the phaeopigments (Phaeo) were calculated using Equations 5.1 and 5.2 respectively given by Lorenzen (1967).

$$\text{Chl } a = \frac{A \times K \times (665_0 - 665_a) \times V}{V \times I} \quad \textbf{(Eq. 5.2)}$$

$$\text{Phaeo } (\text{mg m}^{-3}) = \frac{A \times K \times (R \times 665_0 - 665_a) \times v}{V \times I} \quad \textbf{(Eq. 5.3)}$$

where:

A absorption coefficient of chlorophyll *a* = 11

K factor to equate the reduction in absorbance to initial chlorophyll concentration (2.43)

665_0 absorbance before acidification

665_a absorbance after acidification

v volume of acetone used for extraction (mL)

V volume of mud centrifuged (L)

L path length of the cuvette (cm)

R maximum ratio of 665_0 : 665_a in the absence of phaeopigments (= 1.7).

The calculated chlorophyll *a* values were then divided by 100 and reported in units of mg m^{-2}.

Hepatosomatic index of the mangrove crab *Australoplax tridentata*

Ten male *Australoplax tridentata* crabs were hand-collected from within each site. Only males were collected as the HSI is less affected by the energy demands for reproduction in males than in females (Kyomo 1988; Kennish 1997). All crabs were placed into ice slurry immediately after collection, returned to the laboratory and frozen until analysis.

In the laboratory, the crabs were thawed and dissected by removing the carapace. The hepatopancreas was removed using fine pointed forceps, placed into an aluminium dish and dried to a constant weight in a desiccating oven at 60°C. The remaining crab tissues were placed into a separate dish and dried to a constant weight at 60°C.

After drying, the hepatopancreas and body tissues were weighed (to the nearest 0.01 mg) and the HSI was calculated using Equation 5.4.

$$\mathrm{HSI} = \frac{\text{Hepatopancreas mass}}{\text{Body mass}} \times 100 \qquad \textbf{(Eq. 5.4)}$$

Statistical analysis

All data was analysed using 3-way ANOVA (SPSS version 12.0.1) to test for the effects of the following factors:

- Influx location—(three levels: Coombabah Creek, STP Creek and Helensvale stormwater)
- Influx distance—(two levels: 0 m and 250 m from influx, with one distance replicate on either side of the influx)
- Season—(two levels: wet and dry).

Additional tests such as multiple comparisons conducted are mentioned where appropriate, using the Tukey HSD test. Data were suitably transformed to meet the assumptions for ANOVA before analysis.

Fish and shore bird assemblages

Fish assemblage

During the period 2 May 2005 to 24 June 2005 various fish sampling methods were tried to determine the most efficient method for obtaining representative fish samples from within the intertidal zone of Coombabah Lake. These methods included the deployment of a 33 × 1.5 m monofilament gill net, with 40 mm mesh size, a 10 × 2 m seine net with 1mm mesh and four 700 × 700 mm square hoop fyke nets with 6 mm honeycomb mesh and 4 m wings. Fyke nets appeared to sample the highest number of fish and other nekton species and were used over the following 6 months to sample fish inhabiting the intertidal zone of the lake during high tide.

Four fyke nets were deployed at four random, separate locations around Coombabah Lake on 4 July 2005, 19 August 2005, 28 October 2005 and 3 January 2006. The nets were deployed during low tide with the wings overlapping the mangrove pneumatophore fringe within the tidal zone. On 4 July and 19 August, the nets were deployed during the afternoon low tide and cleared during low tide on the following morning, so that nekton inhabiting the intertidal zone during the evening high tide were sampled. On 28 October the nets were set up during the morning low tide and cleared during the evening low tide, sampling nekton which used the intertidal zone during the daytime high tide. (Very low numbers of fish were caught on 28 October due to very small high tides resulting in negligible inundation of the fyke nets during high tide).

On 3 January the nets were deployed during the daytime and evening low tides as strong northerly winds and large afternoon high tides prevented a typical low tide from occurring at the sampling sites within the lake. This would have reduced the flow of the outgoing tide through the nets, reducing their sampling efficiency. Subsequently, the nets were cleared early the following morning during a more typical low tide. Due to the nature of the sampling gear used and the variable environmental conditions during the sampling trips, the catch collected is meant to provide only a qualitative indication of the fish species present in Coombabah Lake.

On each sampling occasion fish were removed from the nets and placed into an ice slurry, then transported to the laboratory where they were identified to species level.

Shore bird assemblage

Random sites were visited on foot by a single observer during daytime low tide over a 6-month period between August 2005 and January 2006. At each site, the number of each bird species using the mudflats was recorded for each replicate count. Each count comprised of 3 minutes of continual observation and was carried out within an area of approximately 1 ha. The 3-minute observation duration reduced the chances of recounting birds and the roughly 1 ha survey area enabled the observer to reliably identify birds using hand-held 8×21 binoculars.

Results and discussion

Hepatosomatic index of ***Australoplax tridentata***

The body condition of *Australoplax tridentata*, as indicated by the hepatosomatic index (HSI), was significantly better at the 250 m sites at the Coombabah Creek and STP Creek locations during the dry sampling period (Figure 5-2). Values were 4.27 ± 1.08 and 4.19 ± 1.96 (mean ± 1 SD), respectively, at the STP Creek and Coombabah Creek 250 m locations during the dry period, but decreased to 3.10 ± 1.49 and 3.12 ± 1.73 during the wet period.

There was a significant effect of distance from the influx point on the HSI value (ANOVA, $p = 0.004$, Table 5-1). The overall mean HSI at the 0 m and 250 m sites were 3.12 ± 1.97 and 3.57 ± 1.76, respectively, with larger differences demonstrated at the Coombabah Creek and STP Creek locations.

The HSI differed significantly between the influx (0 m) and control (250 m) sites for two of the three locations (STP Creek and Coombabah Creek) and the pattern was opposite in the wet and dry seasons (Figure 5-2). Crabs occurring at 250 m away from local influx sites had significantly higher HSI values during the wet season than their counterparts at the influx site. This difference between 0 m and 250 metre distances is not apparent at the Helensvale stormwater location. This different response to urban influx between wet and dry season is demonstrated by a significant interaction between season and distance from influx (3-way ANOVA, $p < 0.01$) (Table 5-1).

During the wet season there appears to be no effect of distance at any of the sites. This seems to be caused by a reduction in crab HSI in the 250 m sites at Coombabah Creek and STP Creek, as well as a slight increase in crab HSI in 0 metre sites at Coombabah Creek and STP Creek.

This indicator therefore suggests that there is a significant local effect of urban influx on the body condition of *A. tridentata*. This local influence is further modified by rainfall, carrying runoff chemicals farther away from the influx point during the wet period. The influx of urban runoff carried by rain during the wet sampling period greatly reduced the condition of crabs at the control sites (from HSI ~4 to ~3, Figure 5-2). The dry–wet transition had negligible effects, however, on the HSI of the 0 m sites, as these are constantly under the influence of the effects of influx.

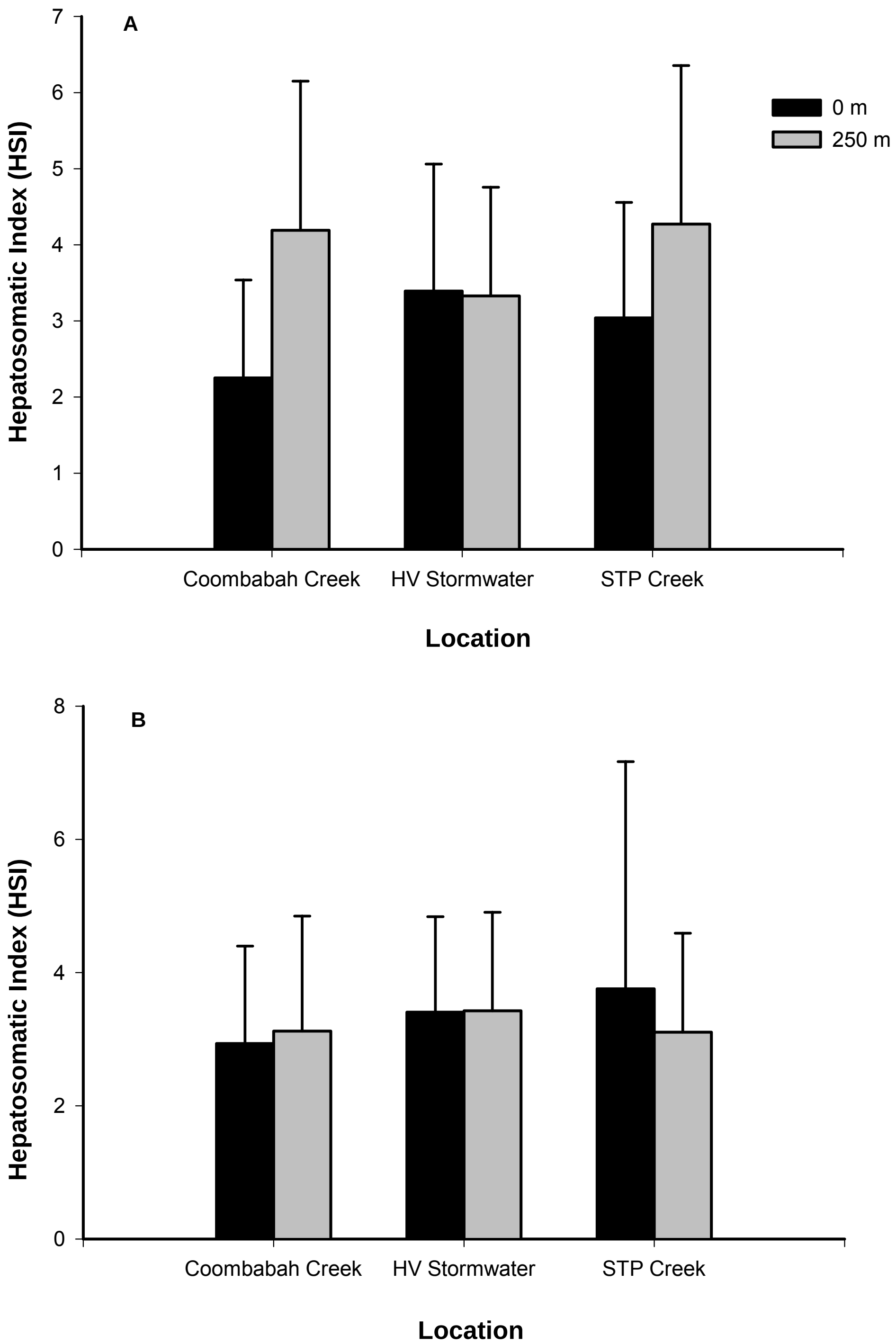

Figure 5-2. Pattern of the mean hepatosomatic index (HSI ± 1 SD) of *Australoplax tridentata* between sites located 0 m and 250 m from urban influx at the Helensvale stormwater outflow, Coombabah Creek mouth and STP Creek mouth, during (A) the dry (12 September – 14 October 2005) and (B) the wet (7 December 2005 – 5 January 2006) sampling periods

Table 5-1. Results of 3-way ANOVA test for differences in mean *Australoplax tridentata* hepatosomatic index (log_{10}-transformed) between sites located 0 metres and 250 m from urban influx at the Helensvale stormwater outflow, Coombabah Creek mouth and STP Creek mouth, during wet and dry seasons (12 September – 14 October 2005 and 7 December 2005 – 5 January 2006, respectively). Significant factors are highlighted in bold.

Source	Type III sum of squares	df	Mean square	F	p
Corrected model	.927[a]	11	.084	2.74	.002
Intercept	130.250	1	130.250	4225.30	<.001
SEASON	.011	1	.011	.357	.551
LOCATION	.102	2	.051	1.65	.194
DISTANCE	.253	1	.253	8.21	.004
SEASON * LOCATION	.036	2	.018	.59	.555
SEASON * DISTANCE	.228	1	.228	7.40	.007
LOCATION * DISTANCE	.174	2	.087	2.82	.061
SEASON * LOCATION * DISTANCE	.124	2	.062	2.00	.136
Error	10.730	348	.031		
Total	141.900	360			
Corrected total	11.660	359			

[a] R^2 = .080 (adjusted R^2 = .050)

During the dry season, the mean crab HSI is significantly lower in sites located 0 metres from urban influx than in sites located 250 metres from urban influx at Coombabah Creek and STP Creek. This difference between 0 m and 250 m distances is not apparent at the Helensvale stormwater location.

These responses in crab HSI are a reflection of the fact that influx at Coombabah Creek and STP Creek was more widely dispersed during the wet season than during the dry season. A total rainfall of 14.6 mm was recorded for the Gold Coast between 1 September and 14 October, 2005, whereas 126.2 mm of rain fell in this region between 1 December 2005 and 5 January 2006 (Bureau of Meteorology 2006). The increased rainfall during the wet season most probably caused an increased influx of catchment urban runoff to the lake at the Coombabah Creek mouth and the STP Creek mouth. This runoff is likely to have lowered salinity, increased turbidity and delivered a cocktail of urban pollutants throughout the lake, causing stress on crabs located within the 250 m sites as well as within the 0 m sites.

The absence of a distance effect at the Helensvale stormwater location suggests that the amount of urban influx at this site is negligible. This could be because the stormwater pipe outflow is located approximately 30 m from the mangroves (where crabs were collected) and drains through a network of reeds and salt marshes before reaching the mangroves during flood periods.

The slight increase in crab HSI in 0 m sites at Coombabah Creek and STP Creek during the wet season could have been caused by increased nutrient and organic matter loads and lowered salinities, making conditions favourable for increased algal growth and food availability for crabs within these sites (refer to $\delta^{15}N$ and chl *a* results).

The HSI recorded for *A. tridentata* in this study is generally indicative of a healthy crab population. The values ranged from 3 to >4 in the dry season, and varied narrowly around 3 in the wet period (Figure 5-2). The dry period values compare favourably with those recorded by Lee and Kwok (2002) for two mangrove grapsids, *Perisesarma bidens* and *Parasesarma affinis*, in a polluted mangrove in Hong Kong.

$\delta^{15}N$ signature of *Avicennia marina* leaves

The pattern of the $\delta^{15}N$ signature of leaves of *Avicennia marina* was consistent throughout the dry and wet sampling periods (Figure 5-3). The lack of a significant wet–dry period difference for this medium-term indicator is supported by the results of the ANOVA (Table 5-2, season effect: $p = 0.661$). There were, however, significant differences between the signatures of leaves collected from different locations, with the most enriched signatures recorded from the STP Creek, demonstrating an enrichment of up to 1‰ over the corresponding values at Coombabah Creek (Figure 5-3).

The main factor of location was significant ($p < 0.001$), as was the interaction between location and distance (location * distance, $p < 0.001$) (Table 5-2). This significant interaction results from the fact that while there was no difference between the influx (0 m) and control (250 m) sites at both the Coombabah Creek and Helensvale stormwater locations, significantly more enriched $\delta^{15}N$ values are evident at the influx site in both the dry (4.43 ± 0.76 vs 3.83 ± .95‰) and wet (4.58 ± 0.84 vs 3.64 ± 1.08‰) seasons.

Multiple comparisons suggest that there were significant differences in $\delta^{15}N$ signatures of *Avicennia marina* leaves, with the overall pattern of signatures being Coombabah Creek < Helensvale stormwater < STP Creek (Tukey HSD tests, $p < 0.001$).

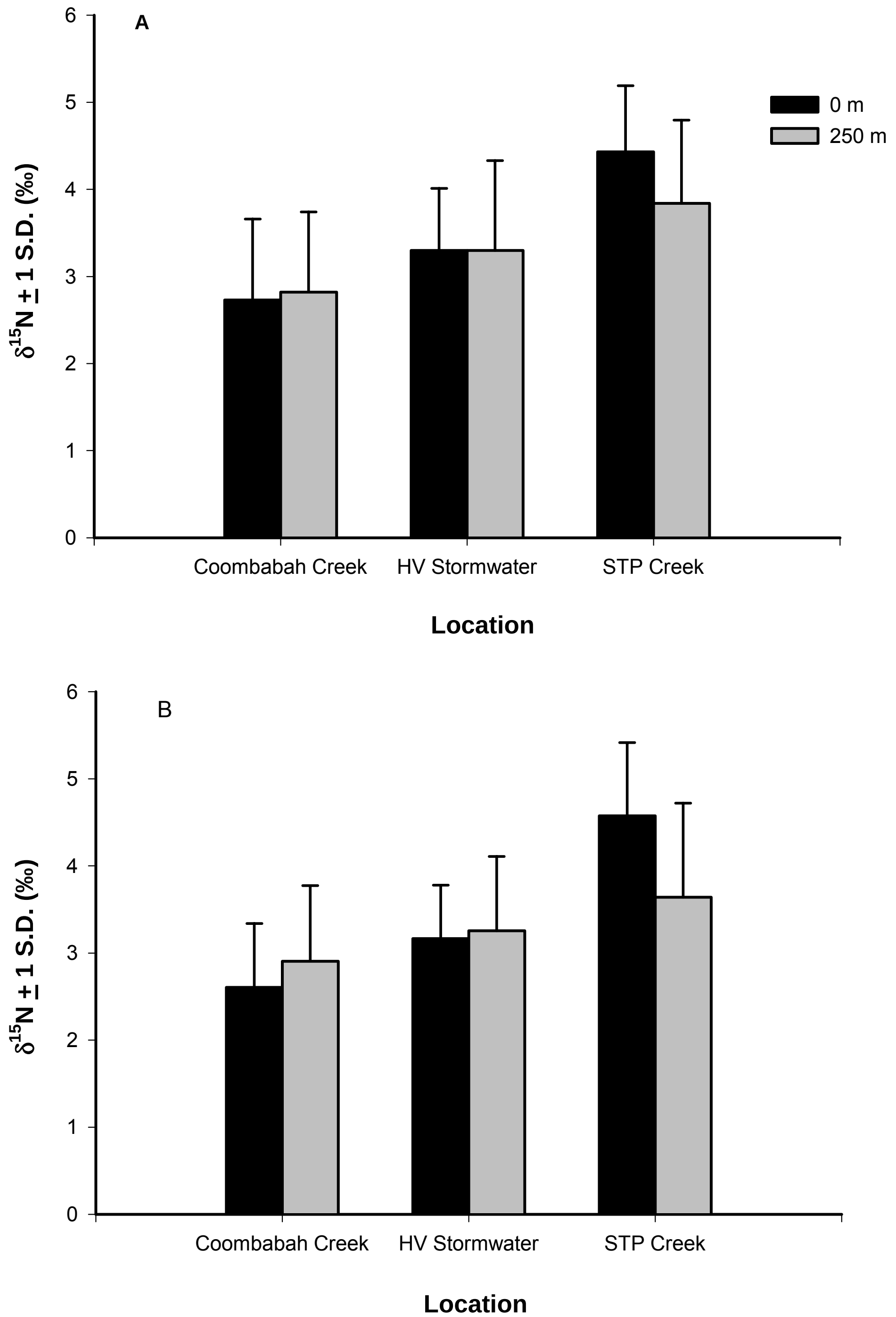

Figure 5.3. Pattern of mean δ^{15}N of *Avicennia marina* leaves between sites located 0 m and 250 m from urban influx at the Helensvale stormwater outflow, Coombabah Creek mouth and STP Creek mouth, during (A) the dry (12 September – 14 October 2005) and (B) the wet (7 December 2005 – 5 January 2006) seasons

Table 5-2. Results of a 3-way ANOVA on the effects of season (wet–dry), location and distance from local influx point on the $\delta^{15}N$ signature of the leaves of the mangrove *Avicennia marina* at Coombabah Lake. Significant factors are highlighted in bold.

Source	Type III sum of squares	df	Mean square	F	p
Corrected model	109.680[a]	11	9.97	13.060	<.001
Intercept	3290.500	1	3290.50	4309.250	<.001
SEASON	.148	1	.15	.190	.661
LOCATION	90.580	2	45.29	59.310	<.001
DISTANCE	2.230	1	2.23	2.920	.089
SEASON * LOCATION	.067	2	.034	.044	.957
SEASON * DISTANCE	.002	1	.002	.003	.955
LOCATION * DISTANCE	12.780	2	6.39	8.370	<.001
SEASON * LOCATION * DISTANCE	1.011	2	.51	.662	.517
Error	219.910	288	.76		
Total	3766.380	300			
Corrected total	329.590	299			

[a] R^2 = .333 (adjusted R^2 = .307)

While the data may reflect a strong contamination by the heavier nitrogen isotope at the STP Creek location, and a stronger urban influx of N at the Helensvale stormwater compared to the Coombabah Creek subcatchment, there is the possibility of a freshwater–marine influence on the observed $\delta^{15}N$ pattern. Water salinity is known to affect nitrogen isotopic fractionation but it is unlikely that a stable strong salinity gradient is present in Coombabah Lake. Further, $\delta^{15}N$ values were considerably higher at the mouth of the STP Creek than at 250 m away, indicating that this local influx source dominated over any marine influence in determining the local $\delta^{15}N$ value. This pattern indicates that there could be some nutrient influx at this site. As the Coombabah STP does not discharge into Coombabah Lake, this enrichment is probably a result of occasional overflow from the treatment plant or underground seepage from pipes.

GHD (2003) obtained similar enrichments in samples of *A. marina* leaves near the STP Creek location ($\delta^{15}N$ = 4.94‰) over the Coombabah Creek location, supporting the occurrence of a local influx of enriched N near the STP Creek location. No mangrove leaves were analysed for the Helensvale stormwater location but the algae *Gracilaria verrucosa* and *Catenella nipae* were analysed, with $\delta^{15}N$ at 4.13‰ and 5.05‰ respectively (GHD 2003). The samples of *A. marina* collected along Coombabah Creek downstream from Coombabah Lake and along Saltwater Creek all returned significantly more enriched $\delta^{15}N$ values ranging from 5.55‰ to 7.29‰, but reducing to 3.68‰ at Paradise Point in the Broadwater (GHD 2003). This pattern further confirms that the enriched $\delta^{15}N$ values near the STP Creek location were not due to a freshwater–marine gradient.

$\delta^{15}N$ signature of the hepatopancreas of ***Australoplax tridentata***

The hepatopancreas of *Australoplax tridentata* was more enriched in $\delta^{15}N$ during the wet (overall mean = 4.18 ± 0.71‰) compared to the dry season (3.26 ± 1.04‰). This pattern seems to apply particularly to the Coombabah Creek and Helensvale stormwater locations (enrichment from $\leq$ 3‰ to $\geq$ 4‰, between the dry and wet periods), while the difference between dry and wet sampling periods was smaller at the STP Creek location (enrichment from 4.15‰ to 4.51‰, Figure 5-4). No general trend is apparent in terms of the effect of distance from the influx sites.

The effect of location was more significant during the dry period, whereas the difference between the influx and control sites at the same location was generally small during this time. Conversely, between-location differences diminished but distance effects became more apparent during the wet sampling period (Figure 5-4).

Mean and standard error plots indicate that mean $\delta^{15}N$ value of the hepatopancreas responds differently in 0 m and 250 m sites over the wet and dry season transition between the Helensvale stormwater (depletion) and Coombabah Creek (enrichment) locations. Values at the STP Creek location sites were essentially equal. There is, however, no significant interaction between distance and season (3-way ANOVA, $p = 0.222$, Table 5-3), even when the data are pooled for all main effects and 2-way interactions ($p = 0.221$).

There is a significant interaction between location and season: during the dry season each location differs significantly (3-way ANOVA, $p < 0.001$) with very little difference in $\delta^{15}N$ between 0 m and 250 m sites.

As with the mangrove leaf $\delta^{15}N$, hepatopancreas $\delta^{15}N$ values are enriched at STP Creek compared to the other two locations and the STP hepatopancreas $\delta^{15}N$ values showed a much smaller response to the wet season than at Helensvale stormwater and Coombabah Creek. It seems that a constant supply of isotopically enriched nutrients enter the lake at STP Creek, having a ‘press’ enriching effect on hepatopancreas $\delta^{15}N$ during both dry and wet seasons, while a “pulsed” supply of isotopically enriched nutrients enters the lake at Coombabah Creek and Helensvale stormwater (during wet season only), having a 'pulsed' enriching effect on the hepatopancreas δ^{15}N at these two locations during the wet season.

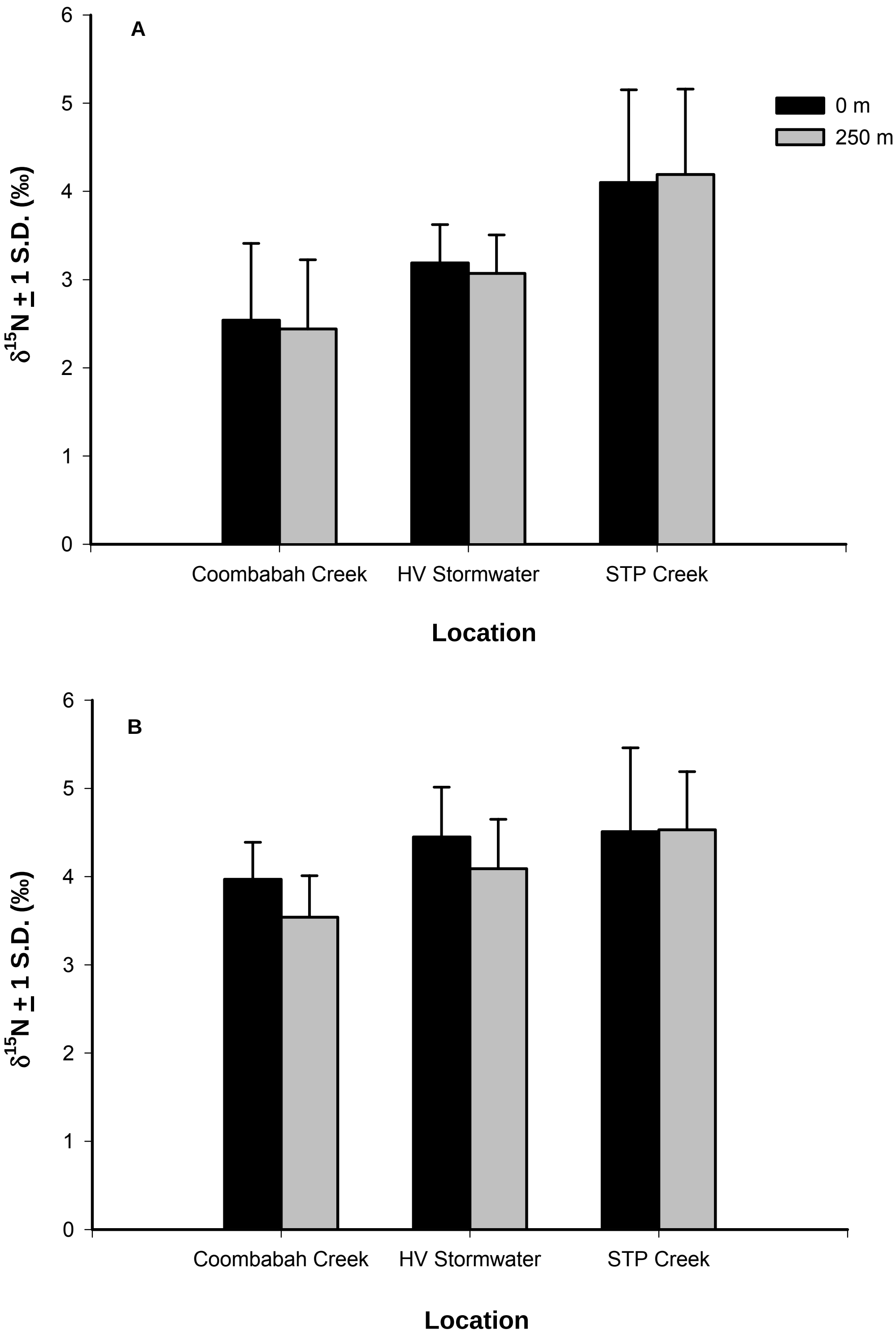

Figure 5-4. The δ^{15}N of the hepatopancreas tissue *of Australoplax tridentata* at the study locations, collected during the (A) dry and (B) wet sampling periods

Table 5-3. Results of a 3-way ANOVA on the $\delta^{15}N$ signature of the hepatopancreas tissues of *Australoplax tridentata* at three study locations in Coombabah Lake. Significant factors are highlighted in bold.

Source	Type III sum of squares	df	Mean square	F	p
Corrected model	160.62[a]	11	14.60	27.14	<.001
Intercept	3980.40	1	3980.40	7397.36	<.001
LOCATION	70.24	2	35.12	65.27	<.001
SEASON	61.35	1	61.35	114.01	<.001
DISTANCE	1.60	1	1.60	2.98	.086
LOCATION * SEASON	11.15	2	5.58	10.36	<.001
LOCATION * DISTANCE	1.53	2	.76	1.42	.244
SEASON * DISTANCE	.81	1	.81	1.50	.222
LOCATION * SEASON * DISTANCE	.23	2	.12	.21	.807
Error	154.97	288	.54		
Total	4258.50	300			
Corrected total	315.59	299			

[a] R^2 = .509 (adjusted R^2 = .490)

During the wet period, a pulsed influx of isotopically enriched N at Coombabah Creek and Helensvale stormwater probably contributes to an enrichment in $\delta^{15}N$ at both of these locations, resulting in $\delta^{15}N$ values similar to those at STP Creek during the wet season. The much lower ratio of sealed surfaces in the subcatchment of STP Creek influx source means that any runoff would be intercepted by the vegetation before entering into Coombabah Lake, thus minimising the effect of rainfall on the $\delta^{15}N$ value at this location.

Also supporting the notion that rainfall during the wet season increased the influx of anthropogenic nutrient into Coombabah Lake is the observation that although there is no significant interaction in $\delta^{15}N$ responses between season and distance, the data indicates a trend that $\delta^{15}N$ values are increased more in 0 m sites at Helensvale and Coombabah Creek during the wet season.

$\delta^{15}N$ signature of the chelal muscle of ***Australoplax tridentata***

The chelal muscle of *Australoplax tridentata* had significantly more enriched $\delta^{15}N$ values than the hepatopancreas tissue collected during the same period. Values were generally >5‰ during the dry period, compared with values between 2.5‰ and ~4‰ for the hepatopancreas. Differences between the two tissue types were reduced during the wet period, but an enrichment of ~2‰ is still evident for all locations.

The general spatial pattern of the $\delta^{15}N$ signature of the chelal muscle of the crab is, however, similar to that of the hepatopancreas, with much more enriched values at STP Creek compared to the other two locations during the dry period, while this difference diminishes during the wet season (Figure 5-5). There were also larger differences in $\delta^{15}N$ between the 0 m and 250 m sites at Coombabah Creek and Helensvale stormwater, but the difference is negligible at STP Creek during the dry season (Figure 5-5).

In the dry sampling period, values at Coombabah Creek and Helensvale stormwater averaged 4.93‰ and 5.29‰, respectively, >1‰ more depleted than the STP Creek average. Again, values were relatively unchanged during the dry–wet transition for the STP Creek location, but significant enrichment at the other two locations now brings the values to ~6‰, close to the 6.6‰ average at the STP Creek location (Figure 5-5). However, $\delta^{15}N$ did become more different between the 0 m and 250 m sites at STP creek (Figure 5-5).

During the transition to the wet sampling period, the enrichment in $\delta^{15}N$ is greater in the 0 m sites than in the 250 m sites at Helensvale stormwater and STP Creek. However, at Coombabah Creek, the increase in $\delta^{15}N$ is greater in the 250 m sites than in the 0 m sites.

Results of the ANOVA do not support a significant effect of distance from the influx point on signature, but a significant effect of both season and location (Table 5-4). The three locations were different from each other, with the signatures being Coombabah Creek < Helensvale stormwater < STP Creek (Tukey HSD test, $p \leq 0.001$), repeating the pattern already established for other parameters based on $\delta^{15}N$ values. Different locations responded to the dry–wet transition in different ways (a significant location * season interaction), and distance from the influx point also did not produce a consistent pattern (significant location * distance interaction). Unlike the hepatopancreas data, there was a significant 3-way interaction ($p = 0.008$), that is, there was more variability in how the chelal muscle signatures changed during the dry–wet transition.

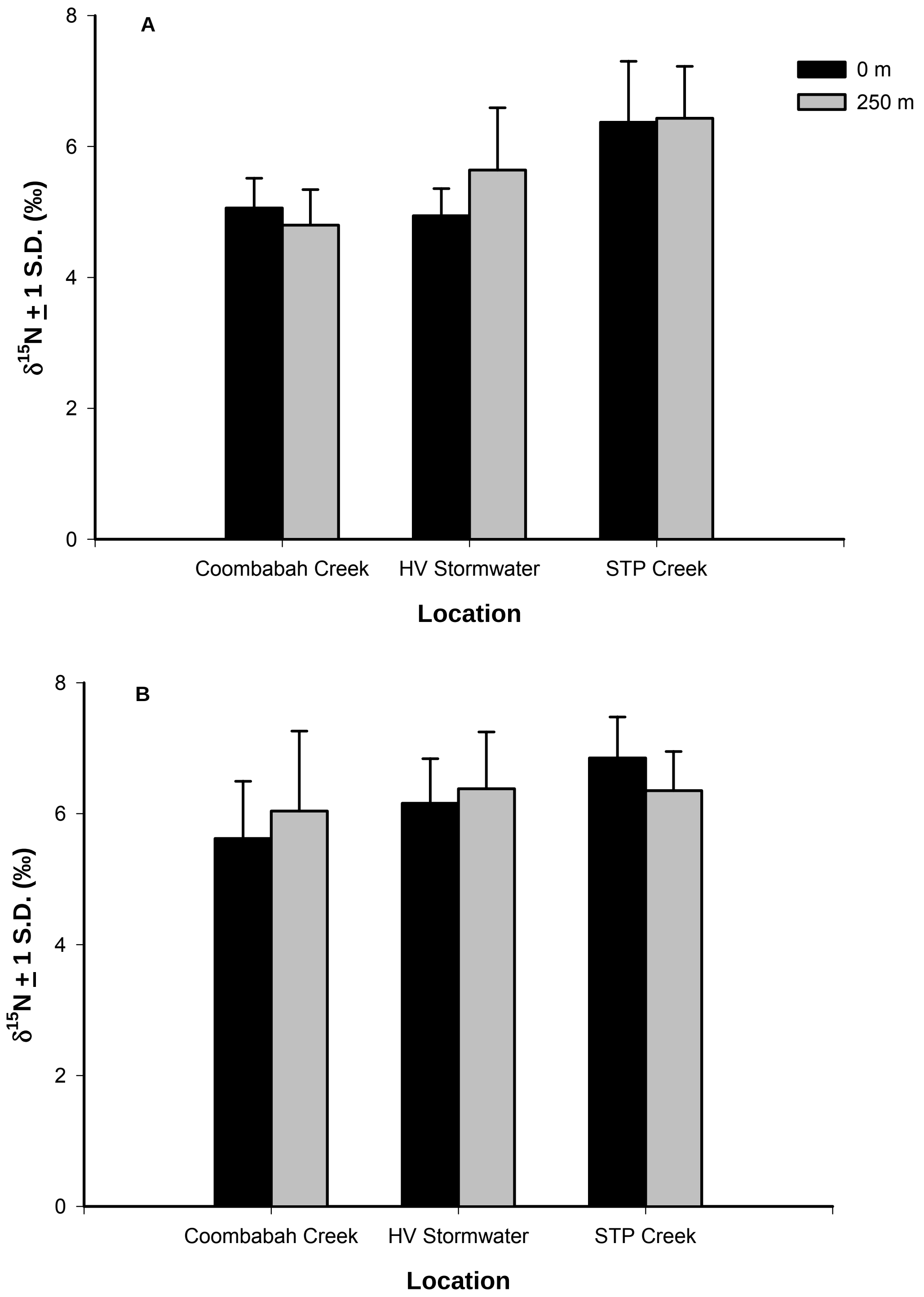

Figure 5-5. The δ^{15}N of the chelal muscle tissue of *Australoplax tridentata* at the study locations, for crabs collected during the (A) dry and (B) wet sampling periods

Table 5-4. Results of a 3-way ANOVA on the $\delta^{15}N$ signature of the chelal muscle tissues of *Australoplax tridentata* at the three study locations in Coombabah Lake. Significant factors are highlighted in bold.

Source	Type III sum of squares	df	Mean square	F	p
Corrected model	129.439[a]	11	11.767	20.011	<.001
Intercept	9977.665	1	9977.665	16967.431	<.001
LOCATION	61.426	2	30.713	52.229	<.001
SEASON	34.583	1	34.583	58.811	<.001
DISTANCE	.806	1	.806	1.371	.243
LOCATION * SEASON	8.802	2	4.401	7.484	.001
LOCATION * DISTANCE	5.481	2	2.741	4.661	.010
SEASON * DISTANCE	.238	1	.238	.405	.525
LOCATION * SEASON * DISTANCE	5.756	2	2.878	4.894	.008
Error	169.358	288	.588		
Total	10448.880	300			
Corrected total	298.797	299			

[a] R^2 = .433 (adjusted R^2 = .412)

The $\delta^{15}N$ values of the crab chelal muscle tissue are similar to those recorded by Lee (2000) for two mangrove grapsids, *Perisesarma bidens* and *Parasesarma affinis*, in a eutrophic estuary.

Coombabah Creek and Helensvale stormwater are expected to deliver larger volumes of catchment influx to the lake during heavy rainfall events. As the main local source of enriched N, the relatively weak stormwater flow at the STP Creek location means that the enrichment in $\delta^{15}N$ would be small during the dry–wet transition. The crab tissue indicators both reflect overall more enriched signatures during the wet season, mainly due to disturbances occurring at the Coombabah Creek and Helensvale stormwater locations. Concomitant to this change at these locations was the signatures of the 0 m sites becoming more enriched than those at the 250 m sites during the wet season (Figures 5-4, 5-5).

Chl *a* concentration in surface sediments

The concentration of chl *a* in the surface sediment of Coombabah Lake generally fell within the range expected of subtropical intertidal mudflats. Yang *et al.* (2003) monitored chl *a* concentration in the intertidal surface sediments of a eutrophic embayment in Hong Kong and recorded peak values at 47 mg m^{-2} during the dry winter period. Chl *a* concentration was generally higher during the dry period in Coombabah Lake (overall mean of 29.4 ± 25.8 and 25.2 ± 24.9 mg m^{-2}, respectively, for the dry and wet periods, Figure 5-6).

The data reveal that this difference mainly arose because of the decrease in chl *a* abundance during the dry–wet transition at the STP Creek location (from 23.8 to 14.4 mg m^{-2}, Figure 5-6). The values were spatially variable, both among locations and among sites within locations, as are reflected by the large standard deviations. The 250 m site on the seaward side of the Helensvale stormwater location had consistently higher chl *a* concentrations compared to the other sites, with an overall mean of 54.2 ± 44.9 mg m^{-2}.

Analysis of the $\log_{10}$-transformed values suggests significant effects of season, location and distance from influx point on sediment chl *a* concentration (Table 5-5), with a significant interaction between season and location. The Helensvale stormwater location (overall mean = 42.7 mg m^{-2}) had significantly higher chl *a* concentrations than did the other two locations (Coombabah Creek: 20.2 mg m^{-2}; STP Creek: 19.0 mg m^{-2}; Tukey HSD test, $p < 0.001$).

The anomaly at the 250 m sites at Helensvale stormwater is probably related to the difference in substrate texture: the two 0 m sites consist of gravel and mud; the ‘upstream’ (right-hand side) 250 m site is fine silt and mud; and the ‘downstream’ (left-hand side) 250 m site is firmly packed, coarse sand and mud. This difference is confirmed by the findings in the section on sediment characteristics, detailed in Chapter 2 of this report. Further, the ‘downstream’ 250 m site appears to have more exposure to sunlight during low tide—due to its being firmly packed and better drained— and the incoming tide would not inundate it as readily as the other three more water-saturated sites.

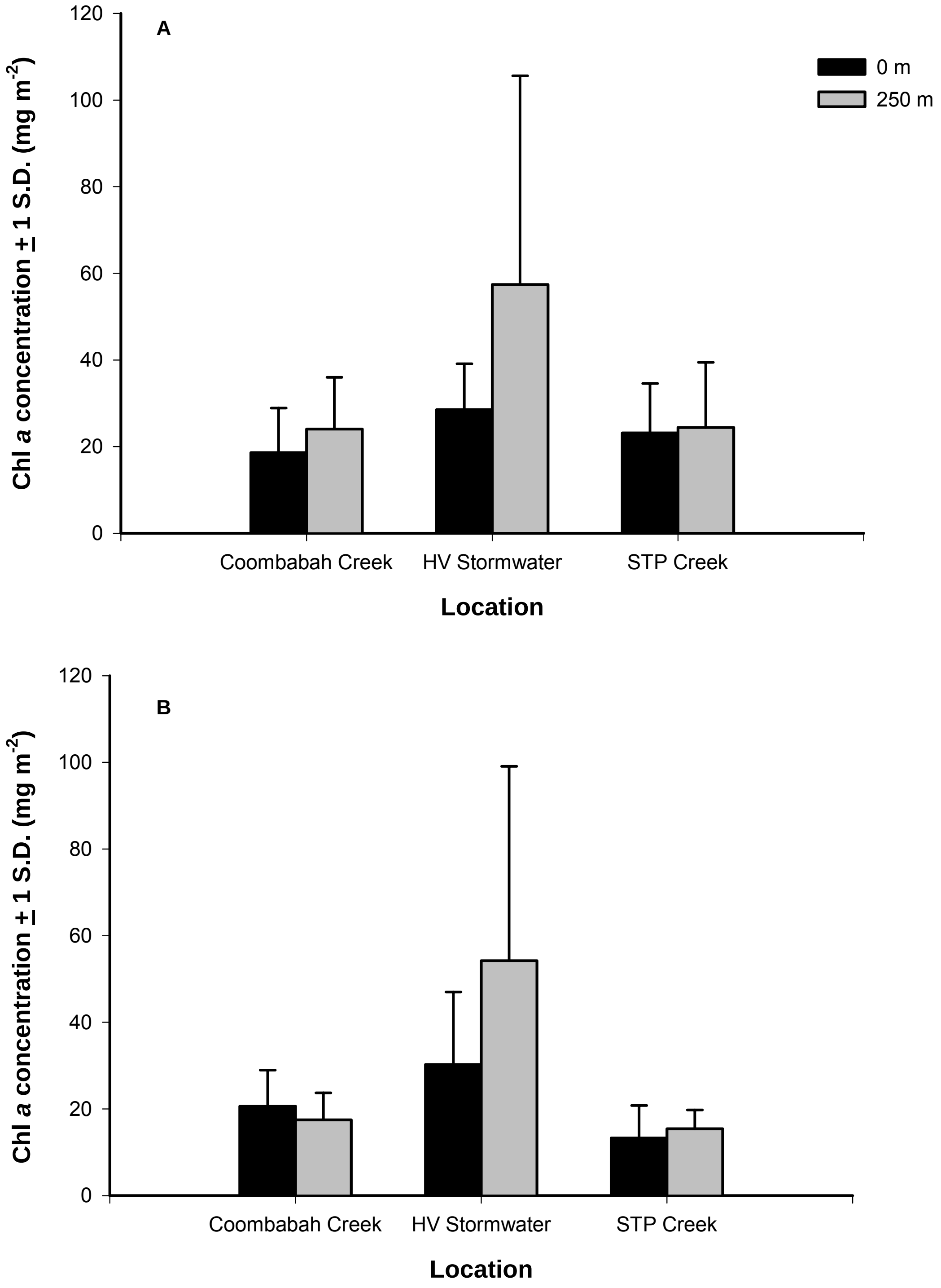

Figure 5-6. The abundance of microphytobenthos (MPB) as measured by chl *a* concentration in surface sediments at the study locations, collected during the (A) dry and (B) wet sampling periods

Table 5-5. Results of a 3-way ANOVA on the chl *a* concentration (log_{10}-transformed) of superficial sediment at the three study locations in Coombabah Lake during the wet and dry sampling periods. Data were log_{10}-transformed before analysis. Significant factors are highlighted in bold.

Source	Type III sum of squares	df	Mean square	F	p
Corrected model	8.485[a]	11	.771	12.785	<.001
Intercept	628.570	1	628.570	10418.601	<.001
SEASON	.596	1	.596	9.878	.002
DISTANCE	.537	1	.537	8.902	.003
LOCATION	6.072	2	3.036	50.322	<.001
SEASON * DISTANCE	.057	1	.057	.948	.331
SEASON * LOCATION	.587	2	.293	4.862	.008
DISTANCE * LOCATION	.234	2	.117	1.942	.145
SEASON * DISTANCE * LOCATION	.340	2	.170	2.817	.061
Error	20.814	345	.060		
Total	657.760	357			
Corrected total	29.299	356			

[a] R^2 = .290 (adjusted R^2 = .267)

The actual stormwater pipe at Helensvale stormwater is located approximately 30 m above the mangrove/mudflat zone where chl *a* samples were collected, and flows into a closed creek surrounded by reeds and saltmarsh. However, observed stream-flow marks through the vegetation after flood periods indicate that the stormwater creek has an ephemeral connection to the lake through the mangroves and mudflats during times of flood. This could cause a pulse response in indicators during the wet season.

Detecting the impact of urban influx on the lake

Five biotic indicators have been used to detect the impact of urban influx on Coombabah Lake, each with different abilities for time integration and responsiveness to components of urban influx, such as increase in sedimentation, anthropogenic nutrient sources and toxic pollutants. The $\delta^{15}N$ signature of the chelal muscle of *Australoplax tridentata* and the leaves of the mangrove *Avicennia marina* are expected to provide a reflection of the condition of the lake with integration over relatively longer time scales compared to that provided by the hepatosomatic index (HSI) and the $\delta^{15}N$ signature of the hepatopancreas. The abundance of the microphytobenthos (MPB), as measured by the concentration of sediment chlorophyll *a*, gives the most rapid response to changes in local condition.

One obvious result generated by these indicators is the difference in conditions among the three influx points. All $\delta^{15}N$ indicators show enrichment along the Coombabah Creek → Helensvale stormwater → STP Creek direction. This pattern could reflect the difference in contribution of anthropogenic N sources among the influx sources, as urban runoff is usually more enriched than natural waters in its $\delta^{15}N$ content (Macko & Ostrom 1994; Cloern 2001; Costanzo *et al.* 2001). Treated domestic sewage has enriched $\delta^{15}N$ values close to 10‰ (McClelland *et al.* 1997). The STP Creek influx point has a much lower human population density in its catchment compared to the other two influx points, as most of the land is still preserved as managed bushland or extensively farmed agricultural land. The fact that this location showed consistently more enrichment than the other locations suggests there is significant local discharge of enriched N here. Some of the possible reasons for this influx of enriched N include overflow or leakage of sewage effluent pipes, or leachate from the Coombabah sludge disposal areas (Sinclair Knight Merz 1997).

Earlier in this report, models have been presented to predict the pattern expected of the indicators, should urban influx from the three point sources have significant impact on the condition of key elements of the ecosystem. The HSI of *A. tridentata* did demonstrate a difference between the 0 m and 250 m sites across all locations as expected, but only in the dry period. Overall there was no significant difference between the wet and dry samples. HSI is a relatively long-term indicator; the response to the dry–wet transition is therefore expected to be easily masked by the timelag for the effects of influx to occur.

The stable isotope indicators generally showed no difference between the 0 m and 250 m sites during the dry period. A distance effect became apparent for the crab tissue indicators, which are more responsive in time to change of conditions in the water column, rendering them more sensitive indicators for urban pollutants carried into Coombabah Lake during the wet period compared with the signatures of mangrove leaves. There was also a significant difference between the wet and dry season values overall for the crab tissue signatures.

Again, this difference was not recorded by the mangrove leaf samples, which tend to give integration of conditions over significantly longer time scales. Mangrove signatures are also more reflective of sediment rather than water column N input. N influx from urban sources is probably more associated with the water column during the early phases of runoff events. Similar to the HSI, the mangrove leaf signature therefore provided a less sensitive indicator to disturbances in nutrient regime in Coombabah Lake resulting from urban input during storm events or diffused occurrences of rainfall. The apparent discrepancy between the indications given by the two crab hepatopancreas indicators is probably a result of the fact that the $\delta^{15}N$ value of the tissue represents only qualitative changes in source of N obtained by the crabs. These changes may bear little correlation with changes in the HSI in time, the variation of which represents changes in the food reserves of the crab.

Data on the stable nitrogen isotope analyses are generally in agreement with each other (refer also to Chapter 2 for corresponding sediment $\delta^{15}N$ values), with a clear spatial trend of increasing enrichment from Coombabah Creek to Helensvale stormwater to STP Creek. This result is not surprising, as the subcatchments of these three influx sources comprise different intensity of $\delta^{15}N$ enrichment. Coombabah Creek upstream of Coombabah Lake drains a large subcatchment still relatively less developed compared with the built-up areas served by the Helensvale stormwater. The larger subcatchment comprises natural bushland and intensively farmed areas in the upper catchment, also drawing water from a larger area and thus a more diluted runoff resulting from the lower subcatchment approaching Coombabah Lake. Domestic sewage effluents are known to demonstrate highly enriched $\delta^{15}N$ values (Lajtha & Marshall 1994; McClelland *et al.* 1997).

The three stable isotope indicators did differ in the within-location and between-season patterns. While all three had negligible differences between the 0 m and 250 m sites during the dry period (Figures 5-4, 5-5), the 0 m site became significantly more enriched only in crab tissue $\delta^{15}N$ during the wet season at the Coombabah Creek and Helensvale stormwater locations. The leaves of *A. marina* remained unchanged during the dry–wet transition (Figure 5-3). This has resulted in an overall enrichment for the two crab tissue indicators during the dry–wet transition but not for the mangrove leaf values.

This difference in response between the two groups of indicators may be explicable by the different sources of N utilised by the mangrove and the crab. Mangroves rely primarily on sediment N sources. *A. tridentata*, being a surface sediment feeder, utilises a mixture of microphotobenthos and mangrove-derived micro-particulate organic matter (micro-POM) for food. The general enrichment of crab chelal muscle tissue by ~3‰ over the corresponding mangrove leaf signature agrees with the shift for $\delta^{15}N$ values across one trophic level (McCutchan *et al.* 2003), supporting the dependence of the crab on mangrove leaf organic matter for food. The simultaneous trophic dependence of the crabs on microphotobenthos may, however, be the reason for the lack of concordance between the mangrove and crab tissue signatures during the dry–wet transition. MPB is more strongly dependent on water column N sources than mangroves are, and MPB is thus more responsive to changes in urban N influx during the dry–wet changeover. Any increase in urban N influx enriched with $\delta^{15}N$ during the wet period will therefore affect the MPB and then the crab tissue signature more than that of the mangroves. The differential change of the crab and mangrove $\delta^{15}N$ values between the dry and wet periods therefore further demonstrate an impact of increased urban influx affecting the metabolism of the biotic assemblage during the wet period.

Fish and crustacean assemblages

A total of 22 species of fish have been recorded during the survey, with abundance levels varying over wide ranges. The five most abundant species in the samples included, in descending order, *Ambassis jacksoniensis, Herklosicthys castelnaui, Gerres subfasciatus, A. marianus* and *Liza subviridis* (Table 5-6).

Two of the nine most abundant species recorded by the GHD (2003) study, namely, *Mugil elongatus* and *Nematolosa come,* were not encountered in the present survey. Conversely, six species were caught in this but not in the GHD (2003) survey (Table 5-6). The GHD (2003) survey recorded 28 fish species over a 27-day period in June 2000 using a wide range of sampling gear ranging from traps, gill and seine nets to angling. The use of fyke nets for sampling in the present study could also have contributed to the difference in species assemblage and relative abundance of the fish recorded in the two surveys, as smaller species such as *Ambassis* spp. probably demonstrate stronger susceptibility to fyke netting than do the larger species.

In addition to the effect of sampling gear, the dominance by a different suite of fishes in this survey may also have resulted from considerable temporal variation in the fish assemblage in Coombabah Lake. Ecoutin *et al.* (2005) recorded about 55 species of fish in both the dry and wet seasons from the 566 km^2 Ebrié lagoon in West Africa, but there were considerable differences between the assemblages of the two seasons, with 20 species making up the permanent fish assemblage. Change in salinity between the wet and dry seasons accounts mainly for the difference in species assemblages.

While Coombabah Lake supports a significant number of estuarine fish species common to this part of Australia, the number of species of high commercial importance is small. Only four species (*Acanthopagrus australis, Platycephalus fuscus, Sillago maculata* and *Sillago ciliata*) are considered to be of significant commercial value. These species are, however, relatively rare in the overall fish catch. Nevertheless, this observation does not imply that Coombabah Lake has little fishery value. The survey by GHD (2003) recorded much higher abundances of juvenile fish over the adults, with the former being an order of magnitude more abundant than the latter at most of the sites sampled.

Table 5-6. Fish and crustacean species recorded in the Coombabah Lake survey during the study period

Group	Species	Common name
Osteichthyes	*Acanthopagrus australis*	Yellow-fined bream
	Ambassis jacksoniensis	Port Jackson glassfish
	Ambassis marianus	Ramsay's glassfish
	Arenigobius frenatus	Half-bridled goby
	Arius graeffi	Fork-tailed catfish
	*Atherinomorus ogilbyi**	Ogilby's hardyhead
	Favonigobius exquisitus	Exquisite sand goby
	Gerres subfasciatus	Black-tipped silver biddy
	Herklosicthys castelnaui	Southern herring
	Hyporhamphus regularis aedelio	River garfish
	*Liza subviridis**	Greenback mullet
	Mugil cephalus	Sea mullet
	Platycephalus fuscus	Dusky flathead
	*Pomatomus saltatrix**	Tailor
	*Pseudogobius olorum**	Blue-spot goby
	*Scathophagus argus**	Spotted scat
	Selenotoca multifasciata	Striped scat
	Sillago ciliata	Sand whiting
	Sillago maculata	Trumpeter whiting
	Tetractenos hamiltoni	Common toadfish
	Torquigener pleurosticta	Banded toadfish
Chondrichthyes	*Aptychotrema rostrate**	Long-snout shovelnose ray
Crustacea	*Acetes australis*	Australian paste shrimp
	*Macrobrachium novaehollandiae**	New Holland river prawn
	*Metapenaeus bennettae**	Bay prawn
	*Penaeus esculentus**	Brown tiger prawn
	*Penaeus plebejus**	Eastern king prawn
	Portunus pelagicus	Blue swimmer crab
	Scylla serrata	Mud crab

* Species not recorded in GHD (2003) survey

Four out of the six species of nektonic crustaceans recorded in this survey have not been recorded in the GHD (2003) study. This, however, may not represent true differences in the assemblages as many of the decapod crustaceans were not thoroughly identified in the GHD (2003) report.

Shore bird assemblage

Sixteen species of shore birds have been recorded from the survey, compared to 26 (31 if species that were 'opportunistically sighted' are included) recorded by GHD (2003) (Table 5-7). There are, however, many species that are not common to the two surveys. Nine of the 16 species recorded in this survey were not recorded by GHD (2003), making the total number of species recorded 35 (or 40 if including opportunistically sighted species) based on the two surveys.

Given the short sampling periods and the seasonal occurrence of many shore birds in the Moreton Bay area, the two lists are unlikely to be representative of the overall shore bird fauna supported by Coombabah Lake.

Table 5-7. Shore bird species recorded in the Coombabah Lake survey during the study period

Species	Common name
Anas superciliosa	Pacific black duck
Ardea alba	Great egret
Ardea intermedia	Intermediate egret
*Egretta (Ardea) garzetta**	Little egret
*Egretta (Ardea) sacra**	Eastern reef egret (grey morph)
*Ephippiorhynchus asiaticus**	Jabiru (black-necked stork)
*Haliastur (Milvus) indus**	Brahmney kite
Himantopus himantopus	Black winged stilt
*Limosa lapponica**	Bar-tailed godwit
*Numenius madagascariensis**	Eastern curlew
Pelecanus conspicillatus	Australian pelican
Platalea regia	Royal spoonbill
*Recurvirostra novaehollandiae**	Red-necked avocet
*Threskiornis molucca**	Australian ibis
Tringa stagnatilis	Marsh sandpiper
*Vanellus miles**	Masked lap-wing

* Species not recorded in GHD (2003) survey

Conclusion

This study employed a novel approach to evaluate the impact of urbanisation on the condition of key structural and functional components of Coombabah Lake, using a combination of spatially explicit sampling design and a range of biotic indicators with different degrees of time integration. Local effects of anthropogenic discharges through identified point sources at Coombabah Creek, Helensvale stormwater and STP Creek have been demonstrated using this approach, with the values of different indicators responding to urban influx in accordance with the spatial (difference in runoff quantity and quality among influx locations, and at control and influx points within location) and temporal (dry versus wet period, affecting the volume and concentration of urban runoff) drivers affecting ecosystem condition. The use of the spatially explicit sampling design around the urban influx locations enabled the detection of local impacts of urban influx at the scale of hundreds of metres.

The values of the biotic indicators suggest that Coombabah Lake is overall still in a relatively healthy condition, with the values of most indicators falling within the normal limits of generally healthy ecosystems. The lake's ability to support beneficial ecosystem services is therefore still largely intact. There are strong indications, however, of significant local N input from STP Creek into Coombabah Lake. The fact that significant changes in levels of the indicators were observable along a distance gradient warns that undesirable local impacts exist due to urban influx. The worsening situation during the dry–wet transition as recorded by some of the short-term indicators further support this conclusion. Increases in urbanisation intensity in the catchment of Coombabah Lake will therefore likely result in further deterioration of ecosystem condition.

Benefits and outcomes

One of the most pressing issues confronting planners and managers in the coastal zone is to find a balance between urban development and conservation of natural resources for future sustainable use. In Australia, the need to achieve this balance is even more urgent as the majority of Australians live within 200 km of the coastline, exacerbating the general global trend of rapid urbanisation of the coastal zone. This study demonstrates how the impact of urbanisation may be assessed at the habitat level using selected novel biotic indicators, in conjunction with a suitable spatial sampling design. The indicators used also respond to different time scales, providing different time integration of indications of ecosystem condition. Although only a snapshot approach, the combined use of these indicators provides a relatively cost-effective way of assessing the 'health' of coastal wetland ecosystems.

Further development

The case-study approach of this study can be complemented with a replicated design at the habitat level to provide a more thorough test of the effects of urbanisation on ecosystem structure and function. The same within-lake spatial design can be applied to additional habitats in the region, and the same biotic indicators used for detecting local as well as lake-level differences in condition.

In order to more strongly relate condition to urbanisation intensity, there is the need for an index of urbanisation to be devised, based on some aspects of the changes that are brought about by urbanisation to coastal wetland habitats. One such metric is the percentage of sealed surfaces in a catchment draining into the wetland. In the case of Coombabah Lake, this index may represent the proportion of built-up versus natural surfaces in subcatchments that would contribute to runoff into the lake through Coombabah Creek or the Helensvale stormwater influx sources.

It is desirable that detailed analyses of the actual level of harmful ingredients contained in the urban influx be conducted to provide an indication of the actual level of pollutants causing change in value of the biotic indicators. For example, the body condition of *Australoplax tridentata* could be enhanced by enrichment of nutrients (such as N) and simple organic matter, but adversely affected by toxic metals or persistent organics. Relatively little is known about the dynamics of the constitution and transformation of urban influx, which is likely also affected by the type of urban development in question. The body burden of anthropogenic pollutants in key organisms may provide information useful for ascertaining the mechanism responsible for the observed pattern in condition along perceived exposure gradients.

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Glossary

Accretion (of sediment)	The natural build-up of sediment over time. The sediment is transported in suspension by water flow, eventually settling and being deposited as a layer of solid particles on the bed or bottom of a body of water.
Acid sulfate soils	Soils containing iron sulfides (e.g. pyrite), commonly found in low-lying coastal areas. When exposed to oxygen through disturbance or drainage, the pyrite undergoes chemical reactions to produce sulfuric acid, which is then flushed from the soil into waterways following rain.
Acid volatile sulfides (AVS)	Highly reactive iron sulfide minerals found in soils and sediments. They are soluble in hydrochloric acid (as opposed to iron disulfides such as pyrite that aren't appreciably soluble in hydrochloric acid). AVS are also called monosulfides.
Advection	The horizontal movement of air; also refers to the transfer of heat by horizontal movement of air (i.e. horizontal convection).
Allochthonous (material)	Describes rocks, soil and any other materials found in a place other than where they and their constituents were formed (i.e. transported from elsewhere); as opposed to 'autochthonous', which describes materials found at their place of origin).
Aluminosilicate	A salt which is both an aluminate (acid form of aluminium hydroxide) and a silicate (derived from silicon dioxide).
Anastamosing wetlands	Wetland systems characterised by their connectivity between parts.
Anthropogenesis	The genesis or development of the human species; hence anthropogenic activities or processes are those involving humans.
Bathymetry	Measurement of the depth of bodies of water.
Benthos	The plants and animals that live on the floor of the sea or lakes.
Berm	A nearly horizontal portion of a beach formed by the deposit of material by wave action.
Bioavailability	The ease with which a foreign material is absorbed by living things.
Biochemical oxygen demand (BOD)	The amount of oxygen required by aerobic micro-organisms to decompose the organic matter in a sample of water, such as that polluted by sewage. It is used as a measure of the degree of water pollution (also referred to as *biological* oxygen demand).
Biota	The total animal and plant life of a region, or sometimes a period, as seen collectively and interdependently.
Biotic indicators	A number of 'typical' or critical plant or animal species whose characteristics can be assessed to determine the condition and trend of a particular physical environment.
Chelal muscle	The tissue in the nipper or claw of crabs and some other arthropods.

Cyanobacteria (micro-MPB)	Unicellular or filamentous photosynthetic organisms, containing chlorophyll and accessory pigments, which thrive in fresh water and contribute to bluish-green scum in late summer; no longer considered true algae.
Ecosystem	A dynamic complex of plant, animal, fungal and micro-organism communities and their associated non-living environment interacting as an ecological unit.
Ecosystem services	The role, functions and 'benefits' provided by particular physical environments such as wetlands, forests, etc. Such services might include the provision of habitat and food supply for flora and fauna, the moderation of water quality, erosion control, recreational amenity and/or aesthetic value.
Ecotone	A transitional zone between two communities containing characteristic species of each. Wetlands are an example, as they are biologically diverse and productive transitional areas between land and water.
Eutrophication	The degradation of water quality due to enrichment by nutrients, primarily nitrogen (N) and phosphorus (P), which results in excessive plant (principally algae) growth and decay.
Fetch (tidal)	The distance travelled by waves with no obstruction, such as the tide washing the shore; affected by the prevailing winds.
Flocculation	Process whereby the attractive forces between clay particles are greater than the repulsive forces, resulting in the formation of larger aggregates of particles. (The converse process is dispersion.)
Fyke net	A long, bag-shaped fishing net held open by hoops.
Geomorphology	The study of landforms, including their origin and evolution, and the processes that shape them.
Grapsids	A family of crabs that can be readily observed in intertidal zones around the world. They are usually flat in shape, with strong nippers and long legs radiating outward, and a sideways movement.
Hepatopancreas	An organ in crustaceans that is used for the storage of nutrients and the removal of toxins.
Hydrodynamics	The science of fluids (liquids and gases) in motion.
Hyporheic zone	The area under or beside a stream channel or flood plain that contributes water to the stream. Hyporheic flow, also called interstitial flow, is the subsurface flow between the watertable and surface water flow. The source of hyporheic flow can be from the channel itself or the water percolating to the stream from the surroundings.
Impervious area (in catchment)	Mainly constructed surfaces—for example, roads, parking lots, footpaths and rooftops—covered by impenetrable materials such as asphalt, concrete, brick and stone. These materials seal surfaces, repel water and prevent water from infiltrating soils. Soils compacted by urban development are also highly impervious.
Interstitial (water)	Water occupying interstices or small spaces such as pore volumes in rock.
Intertidal zone	That area of coastal land that is covered by water at high tide and uncovered at low tide.

Isotope Any of two or more forms of a chemical element having the same number of protons in the nucleus and, hence, the same atomic number, but having different numbers of neutrons in the nucleus and, hence, different atomic weights.

Littoral current A current that moves parallel to the beach (also known as side or longshore current). It is formed by waves breaking at an angle to the shoreline and pushing water sideways. Generally the larger the surf is, the stronger the littoral current will be.

Microphytobenthos (MPB) Microscopic algae and cyanobacteria whose small size and rapid turnover rate respond quickly to environmental fluctuations (such as stormwater runoff), making them ideal biotic indicators for short-term temporal fluctuations or pulse effects of urban influx.

Morphology The branch of biology that deals with the structure and organisation of living things.

Neap tide A tide that occurs when the difference between high and low tide is least; the lowest level of high tide. Neap tide comes twice a month, in the first and third quarters of the moon.

Nekton The group of living organisms that live in the water column of the ocean and freshwater lakes and can propel themselves independent of the currents in the water mass (i.e. they are active swimmers).

Nephelometry Study of the size and concentration of particles suspended in a turbid liquid (in this case lake water), as measured by a probe.

Nutrient cycling A circuit or pathway by which a nutrient moves through compartments of an aquatic or terrestrial ecosystem. In effect, the nutrient (or other element) is recycled, although in some such cycles there may be places called "sinks" where the element is accumulated or held for a long period of time.

Outwelling Term derived from the ‘outwelling’ hypothesis of Odum (1980), which proposed that marsh-estuarine ecosystems produce excess amounts of materials which are then exported to the nearshore environment.

Particulate matter A collection of (usually small) particles of material. Water-borne particulates (as here) include silt, sand, gravel and other insoluble material, while airborne particulates (usually referred to in association with air pollution) are a complex mixture of extremely small particles and liquid droplets including acids (such as nitrates and sulfates), organic chemicals, metals and soil or dust particles.

Phaeopigment A chemically-degraded derivative of phytoplankton chlorophyll. When phytoplankton die (or are eaten and then excreted in zooplankton faecal matter), the chlorophyll in the living algal cell will be degraded to phaeopigments, principally phaeophorbides a and phaeophytins a.

Phytoplankton Microscopic plants (the plant component of plankton) such as fungi, algae, bacteria and yeasts that live abundantly in oceans, and form the foundation of the marine food chain.

Pneumatophore An erect root that rises up above the soil or water and promotes gas exchange. Pneumatophores, or breathing roots, are formed by plants such as mangroves, since there is little oxygen available to the roots in waterlogged conditions. They have numerous pores or lenticels over their surface, allowing gas exchange.

Riparian zone	Riparian zones typically consist of vegetated corridors adjacent to stream channels. They are an effective natural barrier, providing important sources of organic matter and shade, preventing erosion, acting as temperature regulators and filtering runoff water before it enters a stream.
Runnelling	The construction of wide, shallow, spoon-shaped channels to increase tidal flushing of coastal wetlands; considered an effective technique for mosquito control in urbanised soft-sediment coasts, as, among other things, it alters the wetland community structure on which mosquito survival depends.
Sediment	Any particulate matter that can be transported by fluid or air flow and which eventually is deposited as a layer of solid particles on the surface below. Sediments have different settling velocity, depending on their size, volume and density. Desert sand dunes and loess are examples of wind transport and deposition.
Surficial sediments	Sediments on the surface of a lake or sea bed, or, in a terrestrial context, on the surface of the land.
Suspended sediment	That portion of the total sediment load of rivers or lakes that is carried in the water column. It contains the portion termed "wash load" or that portion of the suspended load not represented in the bed material.
Tidal flushing	The exchange of water in a lagoon during high tide. This exchange removes potentially stagnant water and provides input of sea water, nutrients and sediment, which are important factors in the ecological health of a lagoon.
Tidal inundation	The flooding of an area by the incoming tide.
Tidal prism	The change in the volume of water covering an area, such as a lagoon or wetland, between a low tide and the subsequent high tide; this volume depends on water input from the ocean (tides) as well as freshwater input from surrounding drainage (runoff).
Topography	A term that refers to the "lie of the land", or its physio-geographic characteristics such as elevation, slope, and orientation.
Trophodynamics	The forces at work in the composition and behaviour of the food web in a particular environment.
Turbidity	The cloudiness or haziness of water (or other fluid) caused by the suspension of individual particles that are generally too small to be seen without magnification.
Urbanisation	The replacement of natural habitats by built-up areas that support human inhabitation and its associated areas. It is often referred to as the degree of or increase in urban character of a geographic area.
Water column	A top-to-bottom cross-section of a body of water (ocean, river, lake etc.) plus its sediment particles—something like a rock core sample for water. It includes all the water in the water body other than that at the very surface or at the floor, and is also called the pelagic zone. In contrast, the demersal zone comprises water that is near to (and thus is significantly affected by) the surface conditions or the sea/river bed.
Wetland	Biologically diverse and productive transitional areas between land and water characterised by shallow water overlying waterlogged soil and interspersed with submerged or emergent vegetation.