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**Published**

2007

**Journal Title**

Journal of Coastal Research

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## Short-term Variability of Physico-chemical Parameters and the Estimated Transport of Filterable Nutrients and Chlorophyll-*a* in the Urbanised Coombabah Lake and Coombabah Creek System, Southern Moreton Bay, Australia

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### ABSTRACT

DUNN, R.J.K., ALI, A., LEMCKERT, C.J., TEASDALE, P.R., and WELSH, D.T., 2007. Short-term variability of physico-chemical parameters and the estimated transport of filterable nutrients and chlorophyll-*a* in the urbanised Coombabah Lake and Coombabah Creek system, southern Moreton Bay, Australia. *Journal of Coastal Research*, SI 50 (Proceedings of the 9th International Coastal Symposium), 1062 – 1068. Gold Coast, Australia, ISSN 0749.0208

Extensive urban development has occurred within the Coombabah Creek catchment and surrounds over the last two decades, resulting in concerns about degrading water quality. Water quality was investigated three times during summer and three times during autumn/winter at two sites: within Coombabah Lake and at several km downstream within Coombabah Creek. Physico-chemical parameters, suspended solids, chlorophyll-*a* and filterable nutrients (reactive  $\text{PO}_4^{3-}$ ,  $\text{NO}_2^-$ ,  $\text{NO}_3^-$  and  $\text{NH}_3^-$ ) were measured hourly for 13 hours in order to compare the water quality under different tidal conditions at each site. Bathymetric and hydrological data were also collected, which allowed short-term nutrient loads to be estimated. From these measurements, the net transport of filterable nutrients and chlorophyll-*a* into Coombabah Lake and the intratidal variability of Coombabah Creek and Lake waters were determined. Physico-chemical parameters, suspended solids and chlorophyll-*a* concentrations demonstrated characteristic cyclic variations with the influence of tidal and diurnal cycles apparent. Despite elevated nutrient concentrations, chlorophyll-*a* values indicated an oligotrophic-mesotrophic environment, with concentrations ranging between 0.11-3.53  $\mu\text{g L}^{-1}$ . Maximum concentrations occurred during ebb tides, coinciding with periods of increased solar radiation. Elevated total suspended solids were observed during increased current velocities during low tide periods and greater wind speeds within the lake. Filterable nutrient concentrations and transport estimations also demonstrated tidal influences, with increased concentrations observed during sampled high tide phases, indicating increased inputs of nutrients originating from sources other than the creek and lake. Observed reactive  $\text{PO}_4^{3-}$ ,  $\text{NO}_x$  and  $\text{NH}_3^-$  concentrations exceeded Broadwater sub-region values within the Queensland Water Quality Guidelines (2006).

**ADDITIONAL INDEX WORDS:** *Water quality parameters, suspended solids, tide cycle, estuarine system*

### INTRODUCTION

Coastal wetlands and estuaries are spatially diverse systems providing important habitats for flora and fauna including migratory and indigenous fish and bird species in addition to often supporting commercial and recreational fisheries (STUMPF and HAINES, 1998). They also act as filters for contaminants and sediments helping to moderate water quality (FAULKNER, 2004), absorb wave energy and provide cultural and recreational benefits (LEE *et al.*, 2006). However, coastal wetlands and estuaries are under pressure from rapidly-increasing urban populations in coastal areas (PAUCHARD *et al.*, 2006). Continued population growth within coastal regions ensures there will be ongoing impacts on coastal wetland ecosystems (LEE *et al.*, 2006). Within coastal waters, the primary point of entry for nutrients is from terrestrial sources (PEREIRA-FILHO *et al.*, 2001) and for this reason, high primary productivity and biological abundance are often observed in these regions (ALONGI, 1998). In densely populated urban regions nutrient supply is greater due to the entry of both domestic and industrial waste and urban drainage (LEE *et al.*, 2006). The increase of nutrient concentrations within coastal

waters can elicit either positive or negative responses in the ecological health of systems, including the alteration of species richness and abundance (FAULKNER, 2004); productivity (NIXON, 1992); and fishing yields (CEDERWALL and ELMGREN, 1980). Anthropogenic inputs of nutrients may lead to excessive eutrophication, especially where the circulation is restricted, such as in bays and coastal lagoons (LIN *et al.*, 2006). Alterations in chemical characteristics and water quality within coastal systems occurs due to changes in biogeochemical flows (PEREIRA-FILHO *et al.*, 2001), as a consequence of modified land uses resulting in various ecological consequences (LEE *et al.*, 2006). Investigations have previously assessed environmental changes caused by anthropogenic inputs of nutrients and organic material (CARMOUZE and VASCONCELOS, 1992) and time- and tide-series observations of physico-chemical, nutrient and sediment parameters (e.g. DITTMAR and LARA, 2001) in coastal systems. However, fewer studies (e.g. PEREIRA-FILHO *et al.*; (2006)) have quantified these entries. Australia's eastern seaboard is constantly changing from demands by populace for growth, coupled to the economics of increased tourism and development, where ~84% of the population lives within the coastal region (LEE *et al.*, 2006).

One such region is the Coombabah Lake-Creek system in southern Moreton Bay, south-east Queensland, Australia, one of the fastest-growing regions in the developed world (SKINNER *et al.*, 1998). The catchment and local surroundings have undergone rapid urban expansion, including waterfront and golf course developments. Coombabah Lake is the largest estuarine lake in southern Moreton Bay covering  $\sim 2 \text{ km}^2$  (GUTTERIDGE *et al.*, 2003). Despite its modest dimensions, Coombabah Lake is ecologically important as it is a valuable fish (*Queensland Fisheries Act 1994*) and migratory bird habitat (Chinese-Australia Migratory Bird Agreement (1974) and Japan-Australia Migratory Bird Agreement (1986)). In addition to being ecologically significant, the lake system is unique within southern Moreton Bay as it behaves as an inverse estuarine lake during summer periods (Benfer, pers. comm.). The objectives of this study were to observe the intratidal variability of water quality parameters (WQP); total (TSS), organic (OSS) and mineral (MSS) suspended solids; filterable nutrients (reactive  $\text{PO}_4^{3-}$ ,  $\text{NO}_2^-$ ,  $\text{NO}_3^-$  and  $\text{NH}_3$ ); and chlorophyll-*a* (chl-*a*) concentrations in the shallow urbanised Coombabah Creek-Lake waters and estimate the transport of filterable nutrients and chl-*a* at the Coombabah Lake entrance. This study provides the first account of physical, biological and chemical intratidal variability and estimates of filterable nutrient and chl-*a* transportation within the Coombabah Creek-Lake system. Such information permits an initial understanding of the system's behaviour that will assist future management decisions in this ecologically and economically important region.

## METHODOLOGY

### Site description

Coombabah Creek (6914320 m N, 536570 m E; Figure 1) is an urbanised moderately impacted (COX and MOSS, 1999) subtropical tidal creek. The creek is  $\sim 17 \text{ km}$  in length, and originates from the Nerang State Forest. The catchment is urbanised with residential, commercial and industrial development and has an area of  $44 \text{ km}^2$ . The creek flows through Coombabah Lake (6912750 m N, 534400 m E; Figure 1), a shallow body of water characterised by fine sediments located in the mid-tidal region of the creek, with urban development positioned to the east and along the southern and western shorelines. Coombabah Creek ultimately discharges into the Gold Coast Broadwater, a vitally important coastal system both economically and recreationally within southern Moreton Bay. The depth of the creek ranges from  $\sim 3$  to  $< 1 \text{ m}$ , and the width varies between  $\sim 90$  to  $\sim 200 \text{ m}$ . With the exception of shallow channels, the lake is characterised by a relatively flat bathymetry (LEE *et al.*, 2006). Depth within the lake ranges from typically  $< 0$  to  $\sim 1 \text{ m}$  relative to mean water level at low tide, with large portions of the lake becoming exposed during this period. Episodically large inputs of freshwater occur during periods of heavy rainfall, predominantly occurring during summer periods.

### Experimental design

Physico-chemical parameters were monitored in situ and water samples were collected at Coombabah Lake and Coombabah Creek during summer (November, 2005) and autumn-winter sample periods (May-June, 2006). Three sampling events occurred at each site during both seasonal sample periods. Sample collection and in situ measurements were conducted at 1 hour intervals for typically 13 hours at both sample locations, capturing ebb and flood tides. Sampling commenced during the morning

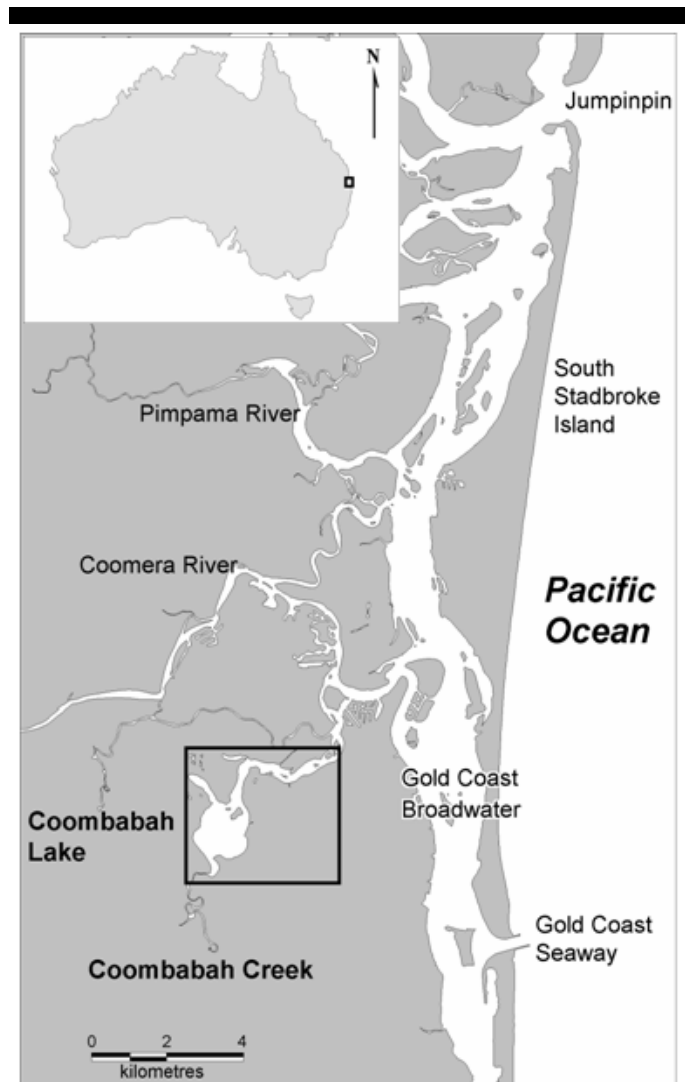


Figure 1. Study area of Coombabah Lake and Coombabah Creek in southern Moreton Bay, Australia (inset)

high tide and incorporated both neap and spring tide periods. Hydrological parameters, conductivity, temperature and turbidity were measured continuously during selected summer lake sampling periods. Hydrological data and water samples were collected at the creek and lake simultaneously during 4/11/05 and 11/11/05. Lake samples were collected in the main channel at the north-eastern entrance (6913140 m N, 535070 m E; Figure 1). This site is  $\sim 160 \text{ m}$  in width and is the only connection between the catchment and creek waters through the lake during low tide. Lake sediments have been characterised in a previous study (LEE *et al.*, 2006). A time lag of three hours (LEE *et al.*, 2006) was used to synchronise tidal observations between high tides occurring at the Gold Coast tidal station and Coombabah Lake-Creek system. An assessment of sample depth was also conducted at the lake entrance during summer sample periods. Water quality parameters, chl-*a*, suspended solids and nutrient concentrations were compared with the Broadwater sub-region values within the Queensland Water Quality Guidelines (QWQG) (2006). Statistical significance was measured at  $\alpha = 0.05$ .

### Water sample collection and analysis

Surface water samples for the determination of TSS, OSS, MSS, chl-*a* and filterable nutrient concentrations were collected ~0.30 m below the water surface using a pole sampler and an acid-washed (10% v/v HCl), sample-rinsed 500 ml low density polyethylene (LDPE) sample bottle (Nalgene). Additional samples from varying depths were also collected at the lake using an adjustable custom-built multi-port sampler. Samples were collected at heights of ~0.06, 0.30 and 1.2 m above the surface sediments through acid-washed tubing and collected by vacuum into acid-washed, sample-rinsed 500 ml LDPE bottles. Furthermore, nutrient and chl-*a* samples were also collected from cross-channel surveys at the lake entrance sample site. These cross-channel sites were nearly equally spaced across, and were used to detect spatial variability of analyte concentrations passing through the sites at any given time. If the concentrations from each of the cross-channel sites were found to be equivalent, then a single mid-channel sample site could be used as a representative sample of all waters passing through the sampled lake site. This cross-channel survey was conducted twice (4/11/05 and 11/11/05). Nutrient samples were immediately filtered through pre-washed, pre-ashed GF/F membranes (25 mm Ø, Millipore) and transferred into 10 ml sterilised polystyrene sample-rinsed tubes. Samples were stored frozen (-20 °C), typically one hour after collection awaiting analysis. Filterable nutrient concentrations were determined by an automated nutrient analyser (Easychem Plus Random Access analyser; Systea Analytical Technologies). Natural filtered seawater references produced by the National Low Level Nutrient Collaborative Trials were used as quality assurance. Recoveries were good – averaging 93.2% for all filterable nutrients from the filtered seawater certified references. Suspended sediment concentrations were determined gravimetrically. TSS samples were filtered through pre-washed, pre-ashed GF/F membranes (47 mm Ø, Millipore), before membranes were air-dried in a desiccator to constant weight (BALLS, 1994). Membranes were afterwards ashed (550 °C, 2 h) for the determination of OSS as loss of weight. MSS were determined as the difference between TSS and OSS. Chl-*a* samples were filtered through pre-washed, pre-ashed GF/C membranes (25 mm Ø, Millipore) immediately after collection. Membranes were then stored frozen in foil-wrapped glass vials. Chl-*a* concentrations were determined spectrophotometrically (665, 750 nm) after acetone extraction and calculations according to LORENZEN (1967).

### In situ water quality parameters

Physico-chemical data were collected at the surface water collection depth. Water temperature (°C), pH, dissolved oxygen (mg L<sup>-1</sup>), salinity and reduction-oxidation (redox) potential (mV) were recorded in situ with a multi-probe analyser (TPS 90-FLMV; TPS Pty. Ltd.), calibrated daily. Turbidity (NTU) was also recorded in situ using a nephelometric turbidity meter (Analite 160; McVan Instruments).

### Field instrumentation

During an extended summer sampling period (31/10/05 to 12/11/05) a submersible sensor base was positioned at the lake entrance sample site consisting of: 1) conductivity, temperature, depth (CTD) and turbidity gauge (NIXIC-CTD-ADC; Falmouth Scientific, Inc.); 2) 3-dimensional open water current meter, Acoustic Doppler Current Profiler (ADCP) (Aquadopp® side-seeing profiler 2 MHz; Nortek AS); 3) high resolution 3-dimensional acoustic Doppler velocimeter (ADV) (Vector fixed stem velocimeter; Nortek AS); and 4) a submersible tide gauge (XR-420-TG; Richard Brancker Research Ltd.). CTD and turbidity

high frequency data were collected using time-averaged data (3.5 minute bursts at a frequency of 10 Hz) obtained every 15 minutes. ADV data collection were obtained using time-averaged data, with burst intervals of 30 minutes at a frequency of 32 Hz and 4096 samples per burst. Submersible tide gauge data were acquired continuously at 1 Hz. Velocity profiles were also undertaken along two Coombabah Creek transects (located approximately 1) 6915290 m N, 537450 m E and 2) 6914360 m N, 536800 m E), using a vessel-mounted ADCP (Workhorse Monitor; RD Instruments) during 4/11/05 and 11/11/05. Water sample collections and in situ WQP measurements were made along these transects. Transects were measured over consecutive ebb-flood tide phases.

### Transport estimations

Transport estimations were determined during summer sample periods at the lake entrance. Cross-channel surveys indicated that concentrations were very similar, permitting the use of a single sample point. Calculations of water volume transport were obtained from current velocity and depth measurements. The area of the entrance cross-section was determined during a bathymetric survey described in LEE *et al.*, (2006). Mass transport values were derived from measurements of water flow and the depth-averaged analyte concentration using the formula:

$$f_i = q_i \times c_i \quad (1)$$

where *f*, *q* and *c* are instantaneous flux, flow and concentration values, respectively. For instantaneous flux calculations, concentrations were converted from µg l<sup>-1</sup> to g m<sup>-3</sup>, then multiplied by the instantaneous water volume transport (m<sup>3</sup> s<sup>-1</sup>) at that time to find flux in g s<sup>-1</sup>. Nutrient concentrations measured along the depth profile were used to provide depth-average concentrations. During this study positive (+) and negative (-) values represent transport out of and into the lake, respectively.

### Meteorological measurements

Meteorological conditions for sampling periods were obtained from a weather station (Australian Bureau of Meteorology station 040764, Gold Coast Seaway) positioned ~7 km south-east of Coombabah Lake. Conditions during the 4/11/05 and 11/11/05 were recorded using a data logging weather station (WeatherMaster 2000; Environdata) positioned within the lake.

## RESULTS AND DISCUSSION

### Hydrological data

Observed tidal levels at the lake entrance displayed mixed tides with a maximum range of 1.00 m during the spring tide and 0.68 m during neap tide periods (Figure 2). Tidal asymmetry was observed with ~5.5 h rising period and ~7.5 h falling period, indicating potential pumping of sediments from downstream Coombabah Creek and other external southern Moreton Bay sources. The lake entrance maximum depth-average velocity observed during flood and ebb tide periods was 0.80 m s<sup>-1</sup> and 0.65 m s<sup>-1</sup>, respectively (Figure 2). Observed lake entrance peak flows were 58.3 m<sup>3</sup> s<sup>-1</sup> and -57.2 m<sup>3</sup> s<sup>-1</sup>. Additionally, ADCP transect data revealed peak flows of 164 m<sup>3</sup> s<sup>-1</sup> and -142 m<sup>3</sup> s<sup>-1</sup> during ebb and flood tide phases at the Coombabah Creek transect sites during the summer sample periods. Higher maximum and lower minimum peak flows within the creek were observed at the downstream transect site compared with the upstream transect location. Seasonal variability of flow volumes with respect to the magnitude of mean flow is to be expected within the lake-creek

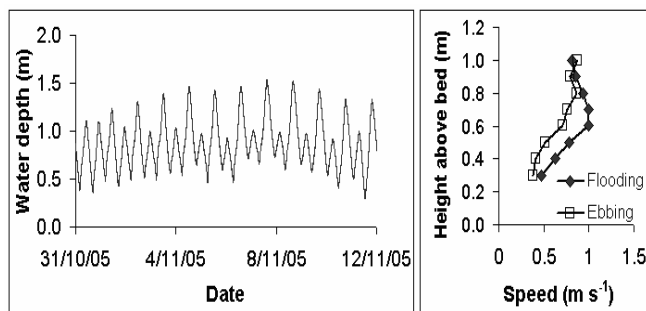


Figure 2. Water depth (left) and sample velocity profiles (right) of the sampled Coombabah Lake entrance channel

system. This current study does not take into account such seasonal events, including periods of heavy rainfall, in which case flow values would be expected to increase significantly.

## Variability within Coombabah Lake and Creek

### Physico-chemical water quality parameters

Surface WQPs measured within the lake-creek system showed characteristic cyclic variations with the influence of tidal and diurnal cycles apparent. Variability ranged between 0.38-151% relative standard deviation (% rsd) (Table 1). Due to the range of most riverine (~7-7.6) and ocean (~8.2) waters, pH values were characterised by the lowest variability. Observed pH and salinity values were similar within the lake-creek system. Mean pH values complied with the Broadwater values within the QWQGs of 8.0-8.4. Water temperature varied due to time of day, water column depth and season, with values ranging from 25.6-29.6 °C during the summer and 15.3-19.6 °C during the autumn-winter sample periods. Maximum temperatures within the lake-creek waters occurred between 1500-1600 h. Dissolved oxygen (DO) concentrations were significantly greater within the creek compared with the lake waters ( $p < 0.001$ ,  $df = 162$ ) (Table 1), possibly being due to a higher oxygen demand within the

shallower lake waters. Low DO levels may be explained by high concentrations of organic matter within the lake sediments, as oxygen is consumed during the consumption of organic matter. DO concentrations exhibited seasonality within the creek with significantly greater concentrations observed during the summer sampling events ( $p = 0.006$ ,  $df = 78$ ). DO concentrations were significantly greater ( $p = 0.003$ ,  $df = 162$ ) during daylight hours compared with night hours, presumably due to the net production of primary producers during hours of solar radiation. DO saturation values within the lake ranged from 53-79%, which is below the lower Broadwater QWQG values. In comparison, the creek DO water saturation values ranged from 62-105%. Mean creek values during the summer (103%) and autumn-winter (80%) periods were comparable with the Broadwater QWQG values of 90-105%. These results support previous findings of COX and MOSS (1999).

### Chlorophyll-*a*

Chl-*a* concentrations within the lake-creek system were comparable with concentrations previously reported in southern Moreton Bay (ABAL and DENNISON, 1996; MOSS and COX, 1999). Mean concentrations were below the Broadwater QWQG value ( $2.5 \mu\text{g L}^{-1}$ ), and suggested an oligotrophic-mesotrophic system based on chl-*a* concentrations (QUINN, 1991). Mean chl-*a* concentrations within the lake and creek waters were not significantly different, nor was there any significant difference in concentrations within the lake and creek waters during neap and spring-tide sampling events. Increased and maximum chl-*a* concentrations were observed during morning ebb tides, occurring during periods of increased solar radiation. Significantly greater concentrations were observed during daylight hours ( $p < 0.001$ ,  $df = 156$ ) compared with samples collected at night-time. Benthic algae associated with mangrove vegetation on the margins of the lake and upper Coombabah Creek flushed during the ebb tide may have contributed to the increased concentrations. Furthermore, the summer period – characterised by longer daylight hours and greater water temperatures – demonstrated significantly greater chl-*a* concentrations within the lake and creek waters compared

Table 1: Minimum (min), maximum (max), mean, standard deviation (sd), and percent relative standard deviation of physico-chemical parameters, chl-*a* and suspended solids within the Coombabah Lake-Creek system ( $n = 42$ )

	Lake					Creek				
	min	max	mean	sd	% rsd	min	max	mean	sd	% rsd
<i>Summer</i>										
pH	7.97	8.09	8.04	0.03	0.38	7.84	8.57	8.10	0.14	2.60
salinity	29.5	36.8	33.2	2.11	6.37	30.5	36.2	34.2	1.68	4.93
redox	42.0	285	182	53.4	29.3	-29.0	158	27.4	42.6	151
temperature	25.6	29.1	27.4	1.04	3.53	26.2	29.6	27.9	0.83	2.72
DO	4.52	6.06	5.46	0.34	6.14	5.94	8.18	6.84	0.49	7.17
chl- <i>a</i>	0.11	3.53	1.48	0.94	63.2	0.11	3.32	1.60	0.92	56.3
turbidity	6.25	36.0	16.53	7.90	47.8	-	-	-	-	-
TSS	93.2	520	227	112	50.4	158	437	286	63.9	22.4
<i>Autumn-Winter</i>										
pH	8.00	8.37	8.12	0.09	1.09	8.01	8.24	8.11	0.07	0.83
salinity	29.6	35.5	33.5	1.86	5.55	30.3	36.2	34.1	2.01	5.88
redox	34.0	280	170	49.6	29.8	-26.0	162	51.3	37.4	74.6
temperature	15.4	19.6	17.8	1.33	17.1	15.3	19.4	17.7	1.24	6.82
DO	5.01	6.12	5.54	0.34	6.12	5.90	7.03	6.58	0.33	4.97
chl- <i>a</i>	0.11	2.14	1.07	0.59	55.0	0.11	2.99	1.05	0.74	70.4
turbidity	6.27	34.38	14.2	6.97	49.0	-	-	-	-	-
TSS	93.8	513	212	104	50.4	27.2	240	127	51.5	40.1

with the autumn-winter concentrations ( $p = 0.039$ ,  $df = 156$ ), suggesting greater phytoplankton biomass during the summer period.

### Suspended solids

Knowledge of suspended sediment dynamics is essential for quantifying fluxes of substances and determining the fate of pollutants (REGNIER and WOLLAST, 1993). Bottom water lake entrance turbidity values ranged from 13.67-403.69 NTU during the summer period and surface waters ranged from 6.25-36.0 NTU, with an average of  $15.4 \pm 7.50$  NTU. Mean turbidity values exceeded the Broadwater QWQG value (6 NTU). Shallow estuarine systems tend to have periodically high turbidity values due to tide- and wind-induced resuspension of bottom sediments and, as such, turbidity is not always a useful health indicator of such systems. Lake turbidity and TSS values significantly correlated ( $r = 0.906$ ,  $p < 0.001$ ,  $n = 83$ ). TSS concentrations ranged between  $93\text{--}520 \text{ mg L}^{-1}$  and  $27\text{--}440 \text{ mg L}^{-1}$  with mean values of  $226.8 \text{ mg L}^{-1}$  and  $202.6 \text{ mg L}^{-1}$  for the lake and creek, respectively. TSS concentrations displayed changes in relation to tidal activity. Increased current velocities and shallower water depths during low tide periods resulted in an increase of TSS concentrations within the lake-creek system, with maximum concentrations occurring during low tide. This was more pronounced within the shallower lake sample site. Turbidity values and TSS concentrations within the system remained relatively constant ( $\sim 50\%$  rsd) with greater variability within Coombabah Creek. TSS concentrations were not significantly different between seasons within the lake-creek system. TSS concentrations were influenced by both tidal action and locally-generated wind waves. Wave heights of 200 mm are commonly observed in Coombabah Lake during strong wind speeds. Maximum TSS concentrations observed at the lake entrance did not coincide with maximum flow values, but rather with increased wind speeds observed during the sample events. The maximum fetch at the lake entrance is  $\sim 1700\text{m}$ , with winds from a south-westerly direction. Northerly winds also create wave actions of interest. During the summer sampling period, increased lake TSS concentrations occurred during periods of elevated wind speeds (11/11/05). Maximum hourly wind speeds ranged from  $8\text{--}30.5 \text{ km h}^{-1}$  from the north-east during this sample period, with an overall mean of  $16.8 \text{ km h}^{-1}$ . Wind speeds greater than  $15 \text{ km h}^{-1}$  were sustained for  $\sim 9.5 \text{ h}$  and coincided with elevated TSS concentrations. Winds from the north with reduced speeds (max.  $15.2 \text{ km h}^{-1}$ , mean  $7.48 \text{ km h}^{-1}$ ) (4/11/05) corresponded with lower TSS concentrations compared with periods of increased wind speeds during the same tide phases. Results suggest that prolonged winds with speeds exceeding  $15 \text{ km h}^{-1}$  from a northerly direction have the ability to directly influence TSS concentrations at the lake entrance and shallower regions. Resuspension events within the lake may impact nutrient cycling (TENGBERG *et al.* 2003). The influence of bioturbation should not be underestimated within the system. OSS and TSS significantly correlated within the system ( $r = 0.990$ ,  $p < 0.001$ ,  $n = 153$ ), with OSS contributing  $9.98 \pm 0.05\%$  to the TSS concentrations under all sample conditions. Organic suspended sediments originate largely from phytoplankton in marine environments; however, no relationship was observed between OSS and chl-*a* concentrations. This is not uncommon in estuarine and coastal environments as part of the OSS is organic detritus originating from terrestrial runoff (PEREIRA-FILHO *et al.*, 2001).

### Filterable nutrients

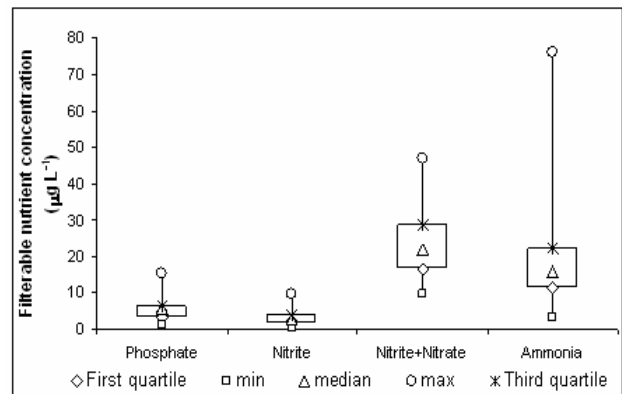


Figure 3. Coombabah Lake-Creek system filterable nutrient concentration box-plot representation

Surface water filterable lake and creek nutrient concentrations were not significantly different through the duration of this study. Reactive  $\text{PO}_4^{3-}$  concentrations within the lake-creek system ranged from  $1.30\text{--}18.2 \text{ µg L}^{-1}$ , with a mean concentration of  $6.98 \pm 4.21 \text{ µg L}^{-1}$  (Figure 3). Elevated reactive  $\text{PO}_4^{3-}$  concentrations exceeded the Broadwater QWQG concentration ( $6 \text{ µg L}^{-1}$ ). Mean  $\text{NO}_2^-$  and  $\text{NO}_3^-$  concentrations within the lake-creek system were  $2.13 \pm 1.80 \text{ µg L}^{-1}$  and  $17.1 \pm 8.76 \text{ µg L}^{-1}$ , respectively. Mean  $\text{NO}_x$  ( $\text{NO}_2^- + \text{NO}_3^-$ ) concentration greatly exceeded the Broadwater QWQG ( $4 \text{ µg L}^{-1}$ ). Maximum  $\text{NO}_x$  concentrations were observed at the lake entrance during high tide phases, with concentrations ranging between  $9.75\text{--}46.8 \text{ µg L}^{-1}$ .  $\text{NH}_3^-$  surface concentrations dominated the lake-creek dissolved inorganic nitrogen (DIN:  $\text{NO}_2^- + \text{NO}_3^- + \text{NH}_3^-$ ) concentrations with a mean contribution of  $37.9 \pm 16.1\%$  and a mean concentration of  $20.6 \pm 9.42 \text{ µg L}^{-1}$ , which exceeded the Broadwater QWQG value ( $8 \text{ µg L}^{-1}$ ). Variability of the surface water concentrations included: 166% rsd for reactive  $\text{PO}_4^{3-}$ ; 188% rsd for  $\text{NO}_2^-$ ; 195% rsd for  $\text{NO}_3^-$ ; and 218% rsd for  $\text{NH}_3^-$ . This was despite the physico-chemical parameters, which have an influence on nutrient concentrations, being much less varied. Several observations indicate that the high variations of filterable nutrient concentrations in the lake-creek system were caused by dynamic processes within the estuarine system, rather than analytical aspects. Firstly, the nutrient concentrations measured were typical of concentrations reported in Australian and local coastal waters (ABAL and DENNISON 1996; COX and MOSS, 1999; MOSS and COX, 1999; WATER ECOSCIENCE, 2003). Secondly, nutrient concentrations plotted against time (Figure 4) demonstrated a pattern with respect to the high and low tide phases. Maximas

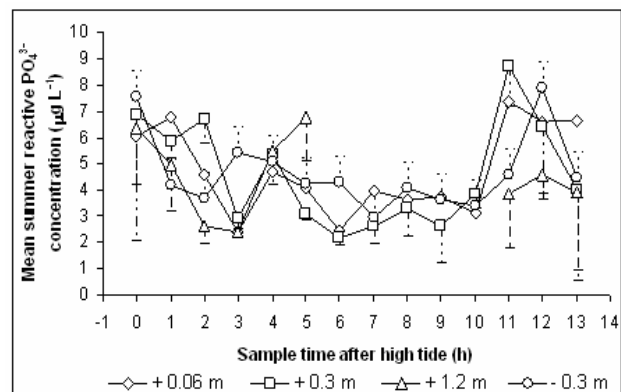


Figure 4. Coombabah Lake filterable reactive  $\text{PO}_4^{3-}$  mean concentrations during summer ebb and flood tide phases

occurred during high water periods and minimas occurred during periods of low water (Figure 4). Therefore, concentrations seem to be responding to natural cycles, and indicate genuine changes. The maximas and minimas observed are likely due to the mixing of two end-member waters with different concentrations, suggesting that the dominant sources of filterable nutrients are external to the lake-creek system. This observation may be explained by the hydrology of the Broadwater. During flood tides, waters entering the lake-creek system originate from the heavily urbanised Paradise Point area and Coomera River. The lower reaches of Coomera River are highly developed, including a number of canal and golf course developments, and upstream land uses include crop growing, dairy farming and cattle grazing. The ebb tide brings waters from Paradise Point and Coomera River past the creek entrance and flood waters, then enter the urbanised creek before entering the lake. Additionally, the initial ebbing waters during high water may also aid in the downstream transportation of potentially elevated concentrations from upstream golf course developments and other catchment nutrient sources. This is admittedly one of several possible interpretations of the results. No significant differences were observed between periods of no rainfall and low recorded rainfall within the lake and creek waters. During heavy rainfall events industrial and commercial development, waterfront housing and impervious surfaces within the catchment and lake-creek foreshores act as point and non-point nutrient sources (GUTTERIDGE *et al.*, 2003). No significant differences in filterable nutrients were identified between seasons within the lake-creek system. Lake  $\text{NO}_3^-$  ( $p = 0.035$ ,  $F = 3.122$ ) and  $\text{NO}_x$  ( $p = 0.022$ ,  $F = 3.510$ ) concentrations demonstrated significant differences between the four sample depths. Mean concentrations showed a general trend of greater concentrations in the surface waters (~0.3 m below the water surface) compared with the deepest sampled waters (~0.06 m above surface sediments). This may be explained by the consumption of filterable nutrients in the overlying waters by benthic planktonic and bacterial communities (ALONGI, 1994).

### Filterable nutrient and chl-*a* transport estimations

Tides are major agents of transport in most coastal environments. The transport of filterable nutrients and chl-*a* at the lake entrance sample site during ebb-flood tide phases were determined using Equation 1. Mean instantaneous nutrient transport loads during the summer sample periods were all negative values, indicating transportation of filterable nutrients into the lake. Lake entrance transport estimations included:  $-0.010 \pm 0.171 \text{ g s}^{-1}$  for reactive  $\text{PO}_4^{3-}$ ;  $-0.014 \pm 0.068 \text{ g s}^{-1}$  for  $\text{NO}_2^-$ ;  $-0.126 \pm 0.569 \text{ g s}^{-1}$  for  $\text{NO}_3^-$ ; and  $-0.152 \pm 0.757 \text{ g s}^{-1}$  for  $\text{NH}_3^-$ . Observed mean flux values for surface waters chl-*a* were determined as  $0.016 \pm 0.066 \text{ g s}^{-1}$ , with a maximum transportation load determined as  $0.167 \text{ g s}^{-1}$ . Estimates were derived solely from directly-measured values of flow and concentration collected across the range of measured flow values encountered during the summer sample periods. These estimations do not detail seasonal variations. The results do, however, provide estimations of filterable nutrients and chl-*a* transportation and their short-term summer variability at the entrance sample site. Flow-averaged mean chl-*a* transport rates were greater during the ebb tide phases at the lake entrance. Additionally, flow-averaged transport rates during sampling events demonstrated greater fluxes during flood tide periods for  $\text{NO}_2^-$ ,  $\text{NO}_3^-$ ,  $\text{NH}_3^-$  than ebb tide periods, and were very similar for reactive  $\text{PO}_4^{3-}$  during ebb and flood periods. Such results indicate that external sources transported nutrients into the lake during the sampling period. Even if the results obtained in this study are not representative of the whole year, they indicate a

potential for eutrophication of lake waters as a result of nutrient input from external sources within southern Moreton Bay. Additionally, waters that enter the lake through a secondary shallower entrance channel during high tide periods would also increase nutrient loads entering the lake system.

### CONCLUSION

Physico-chemical parameters, suspended solids and chl-*a* concentrations within the lake-creek system demonstrated cyclic variations, with parameter variability ranging between 0.38-151% *rsd*. pH and chl-*a* values complied with the Broadwater QWQG sub-region values, and both DO and TSS values failed to abide by the specified guideline values. Tides and winds in excess of  $15 \text{ km h}^{-1}$  were identified as influential in elevating lake entrance suspended sediment concentrations. Filterable nutrient concentrations demonstrated values typically encountered in Australian and local coastal waters. Nutrient concentrations were tidally influenced, with increased concentrations occurring during sampled high tide phases. Reactive  $\text{PO}_4^{3-}$ ,  $\text{NO}_x$  and  $\text{NH}_3^-$  concentrations exceeded Broadwater QWQG sub-region values. Mean negative transport values indicated nutrient inputs from external sources into the lake system during the sample period. Continued assessment is required to provide a better understanding of the hydrodynamics and biogeochemical processes in the ecologically-significant region of Coombabah Lake and Coombabah Creek, which is an important link to material transfer to the Gold Coast Broadwater, southern Moreton Bay.

### ACKNOWLEDGEMENTS

The authors would like to acknowledge the financial assistance of the Cooperative Research Centre for Coastal Zone, Estuary and Waterway Management. Acknowledgments are also made to N. Benfer, D. Dunn, K. Dunn, T. Dunn, C. Scraggs, and P. Williams for their efforts in fieldwork and to M. Jordan for laboratory assistance.

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