

Chapter 4 Nutrient response to flow, water depth, temperature and macrophytes in the Manning River pools.

4.1 Introduction

Along with land use influences, hydrological events, such as low flows and floods, have major consequences to water quality through evaporation, dilution or runoff impacts (Rolls et al. 2012). Agriculturally impacted sites often contribute large amounts of suspended sediment and terrestrial organic material under high-flow conditions. Conversely, low-flow conditions may result in increased nutrients as geological or groundwater influences that became more evident under dry conditions (Williard et al. 2003, Olsen and Hawkin 2012). Increases in dissolved nutrients under summer low flows may occur through physical changes such as water temperatures and stratification in deep pools.

These physical changes may also have ecological impacts through a reduction in habitat, increased of primary productivity and algal biomass (Suren and Riis 2010, Datry 2012). Primary productivity is likely to increase under the warmer temperatures and reduced scouring of summer low flow conditions (Suren and Riis 2010). The increase in primary productivity will alter dissolved oxygen concentrations and in-stream nutrient availability (Neill et al. 2006). Further to this, as periphyton biomass increases and smothers habitats, there may be a corresponding impairment of benthic macroinvertebrate communities resulting in the permanent loss of species that are unable to resist chronic impacts (Datry 2012).

4.1.1 Stratification and nutrient behaviour

Nutrient transformations and transfers within water bodies are influenced by diel O₂ cycles and biological factors such as respiration (Harrison et al. 2005). These influences are altered by physical characteristics, such as water temperature, depth and velocity. Under low flow conditions, saline groundwater influences are also known to contribute to thermal and salinity stratification within streams (Turner and Erskine 2005). The combination of low flows, high solar radiation, increasing water temperatures and salinity influences, commonly results in persistent stratification in lakes and river pools throughout Australia (Western et al. 1996, Turner and Erskine 2005, Oliver et al. 2009, Reinfelds and Williams 2012).

Studies of stratification and the impacts of changes in temperature and dissolved oxygen have mainly focused on large storages or lakes (Szumiec 1985, Boland and Padovan 2002, Wilhelm and Adrian 2008), with very few looking at stratification in rivers (Western et al. 1996, Turner and Erskine 2005). Studies conducted in both lakes and rivers have found that stratification and the resulting compartmentalization of the water column influences heat transfer, the accumulation and transformation of chemical compounds, as well as affecting chemical and biological gradients vertically (Romero and Melack 1996, Eckert et al. 2002, Branco et al. 2009).

An important consequence of compartmentalization is that during stratification, the thermocline restricts the replacement of the oxygen in deeper water layers, increasing the hypolimnetic oxygen deficit, which can lead to anoxic conditions (Eckert et al. 2002, Boehrer and Schultze 2008, Branco et al. 2009) (Fig. 4.1). Deoxygenation is frequently a major outcome from stratification as

the lack of physical mixing results in the consumption of oxygen by benthic organisms in the lower layers without the ability to exchange oxygen with surface layers (Schladow and Hamilton 1995). The anoxic conditions promote the release of nutrients from the underlying sediments due to the creation of a reducing environment (Davis and Koop 2006, Boehrer and Schultze 2008, Branco et al. 2009). The release in nutrients can promote poor water quality and increased primary productivity, frequently resulting in cyanobacterial blooms (Webster et al. 1997, Rahman et al. 2005). Furthermore, the anoxic layer can contain a considerable volume of water which could become unused by aquatic biota (Branco et al. 2009).

When looking at stratification within deep pools in rivers, Turner and Erskine (2005) found that pools with a depth greater than 4 m stratified under low flow conditions resulting in anoxia in the bottom layers. This lack of oxygen resulted in reducing conditions which contributed to order of magnitude increase in total phosphorus beneath the oxycline (where a sharp gradient in oxygen exists) as sediment-bound phosphorus was released, resulting in a general decline in water quality.

Stratification has also been found to result in differences in electrical conductivity between surface and bottom layers (Cavalcante et al. 2007) or stratification may be driven by saline groundwater intrusion under low flows (Turner and Erskine 2005). Saline groundwater intrusion to pools within Australian rivers have resulted in both persistent and episodic stratification (Morrissy 1979, Western et al. 1996, Turner and Erskine 2005). Saline inflows appear to assist stratification development and inhibit mixing through increasing density differences between surface and bottom layers (Turner and Erskine 2005).

4.1.2 Diel influences on nutrient behaviour

Stratification studies that have assessed diel changes within depth profiles (Frempong 1983, Talling 2004, Cavalcante et al. 2007) have found that diel cycles are largely repetitive and usually relate to weather episodes. A study monitoring hourly temperature profiles in a deep natural pool over a 12-month period found that flow had a major influence on the persistence of stratification and cold fronts during summer were effective in increasing mixing, while high wind speeds suppressed the degree of thermal stratification but did not achieve a breakdown in the thermocline (Reinfelds and Williams 2012). These cycles may also extend seasonal stratification by vertical heat transfer through the water column (Tallings 2004). Along with changes in temperature through the depth profile, there is likely to be an associated change in other variables such as dissolved oxygen, pH and conductivity (Cavalcante et al. 2007), as has been found within lake systems. As occurs with stratified lakes, these differences result in changes in nutrient concentrations and forms over a short temporal period (Cavalcante et al. 2007).

Studies that have documented diel oxygen fluctuation in rivers tend to look at interactions between macrophytes and water quality (Grimm and Petrone 1997, Park et al. 2003). Often the changes in diel oxygen cycles relate to plant respiration and photosynthesis influences chemical transformations, particularly those associated with nutrients and trace metals (Brick and Moore 1996). However, there has been relatively little work on how short-term oxygen variations affect nutrient transport through streams (Harrison et al. 2005).

There is also evidence of complete night-time anoxia in river-associated wetlands and stagnant waters (Caraco and Cole 2002, Venterink et al. 2003), while other research has inferred a link between diel fluctuations in oxygen and denitrification rates in a river-associated wetlands (Venterink et al. 2003) and sediments heavily colonized by benthic microalgae (e.g. Christensen et al. 1990, Nielsen et al. 1990). The link between diel patterns in oxygen consumption to patterns of N₂O and N₂ production was also shown by Laursen and Seitzinger (2004). However, few studies have examined the role or implications of night-onset anoxia for nitrogen cycling and transport in streams or rivers. These rapid in-stream fluctuations can affect a broad spectrum of biologically important nutrients over very short timescales (Harrison et al. 2005).

Despite the potential for day-night shifts in oxygen conditions to alter stream chemistry, many models of river biogeochemistry ignore diel changes in stream chemical dynamics (Harrison et al. 2005). Even models that incorporate the ability of rivers to transform nutrients (Fisher et al. 1998; Wollheim et al. 1999), rather than treating them as simple conduits, generally ignore short-term environmental variations despite the fact that diel O₂ fluctuations are relatively common in biologically productive rivers and streams (Harrison et al. 2005).

4.1.3 Nutrient cycling in the Manning River

The Manning River contains pools which exhibit a range of depths and velocities, including macrophyte-dominated shallow pools to frequently thermally stratified deep pools (Thurtell and Bishop 2006). A variety of physical, chemical and biological factors within these pools are thought to influence nutrient availability and cycling (Fig. 4.1); this influence is not currently well understood.

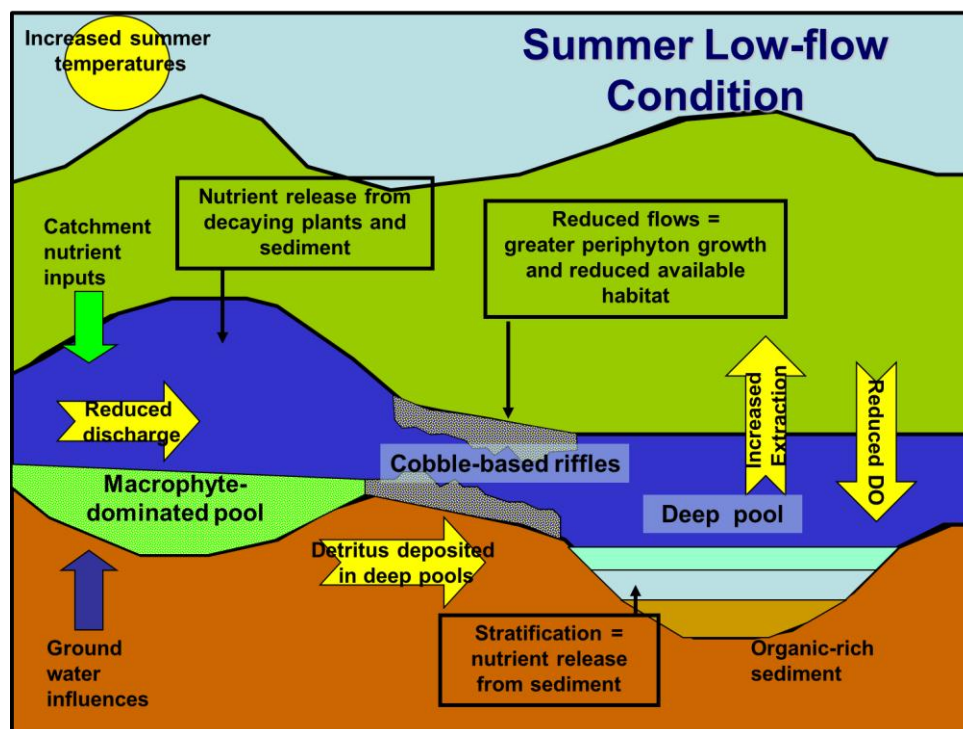


Figure 4.1 Factors influencing nutrient availability and cycling in river pools

Preliminary investigations undertaken in 2007 found strong thermal stratification occurred within two deep (>8m) pools (Ida Lake and Bungay Pool) during the Austral summer (Thurtell 2008) (Figs. 4.2 and 4.3). During the study period it appeared that thermal stratification did not occur at any time during the colder months (April to June), but was occurring at deep pool sites when flows were below $30 \text{ m}^3 \text{ s}^{-1}$ ^b at Killawarra gauge during the warmer months (December to March) (Figs 4.2 and 4.3). It is under these conditions that dissolved oxygen may be depleted in the bottom layers of the deep pool profiles (Thurtell 2008). If stratification is persistent in these deep pools and anoxic conditions exist in the hypolimnion, water quality changes, including increased nutrient release from pool sediments, are likely to occur.

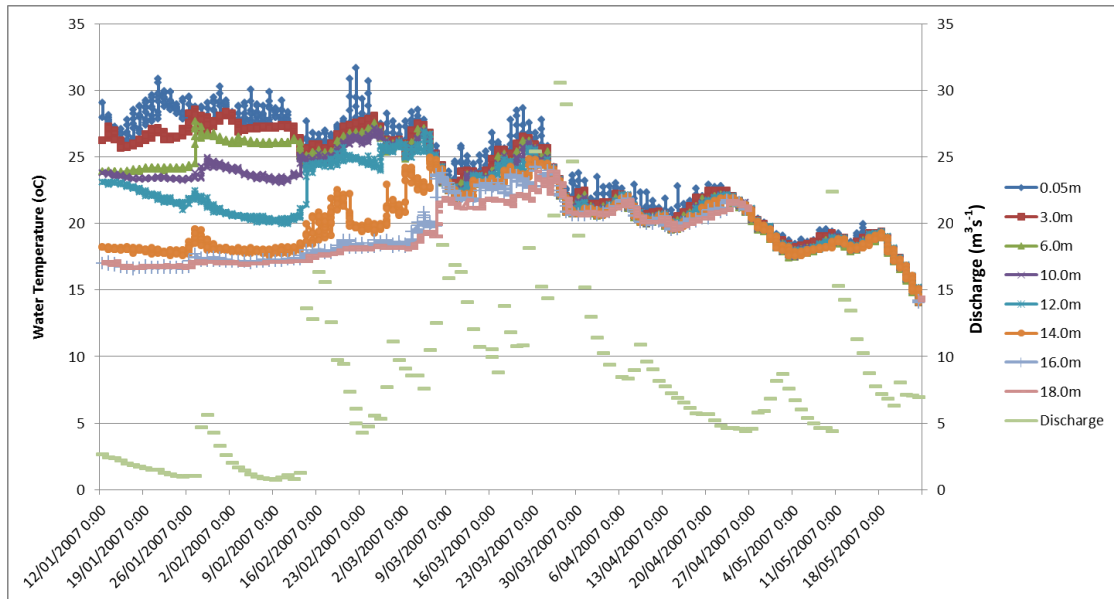


Figure 4.2 Ida Lake water temperature profiles and discharge at Killawarra, January 2007 to June 2007

^b Discharge measured at Killawarra gauging station, ~35km downstream of Ida Lake. (See Figure 4.8)

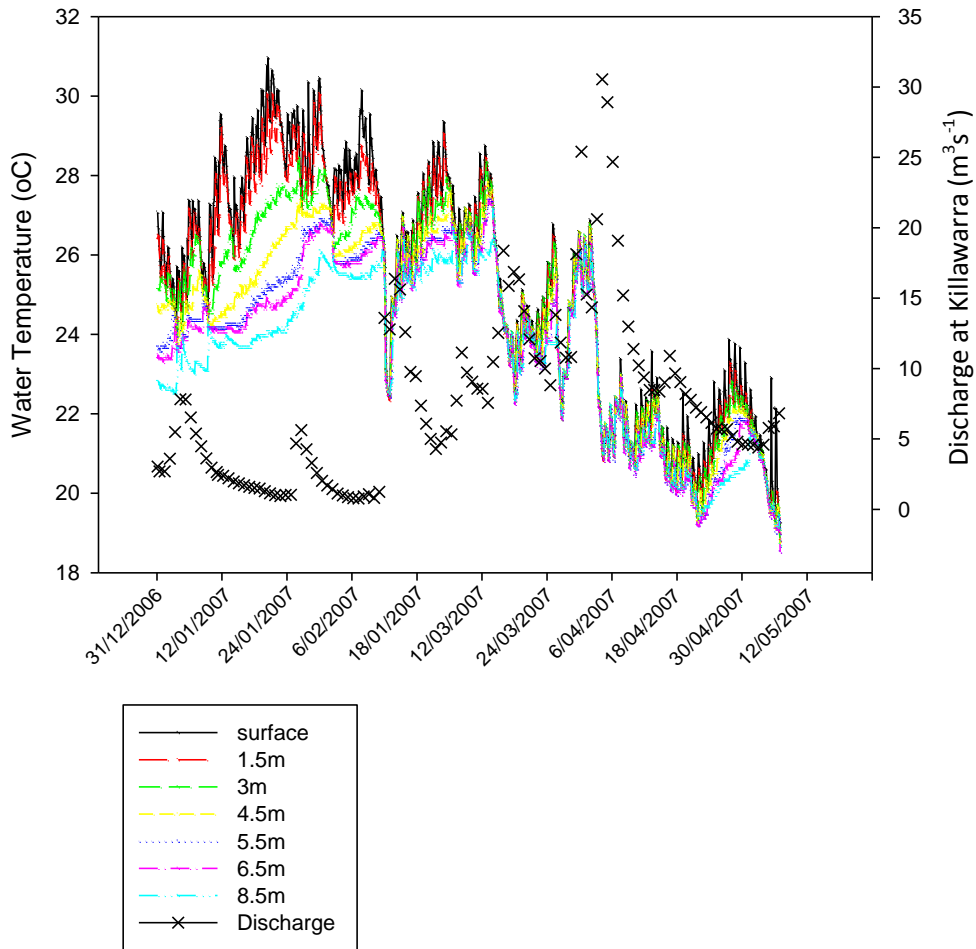


Figure 4.3 Bungay Pool water temperature profiles and discharge at Killawarra, December 2006 to May 2007

As part of the 2007 preliminary investigation, thermistor chains with data loggers were also placed among macrophyte beds in two shallow (<3m) macrophyte-dominated pools (Woodside and Railbridge pools). During January, when flows were around $2.3 \text{ m}^3 \text{ s}^{-1}$ at Killawarra, water temperatures were at least 2°C higher at the surface when compared to epibenthic water temperatures within macrophyte beds (Figs. 4.4 and 4.5). This difference in temperature may have influenced nutrient uptake by macrophytes and rates of release from sediment (Howard-Williams et al. 1989).

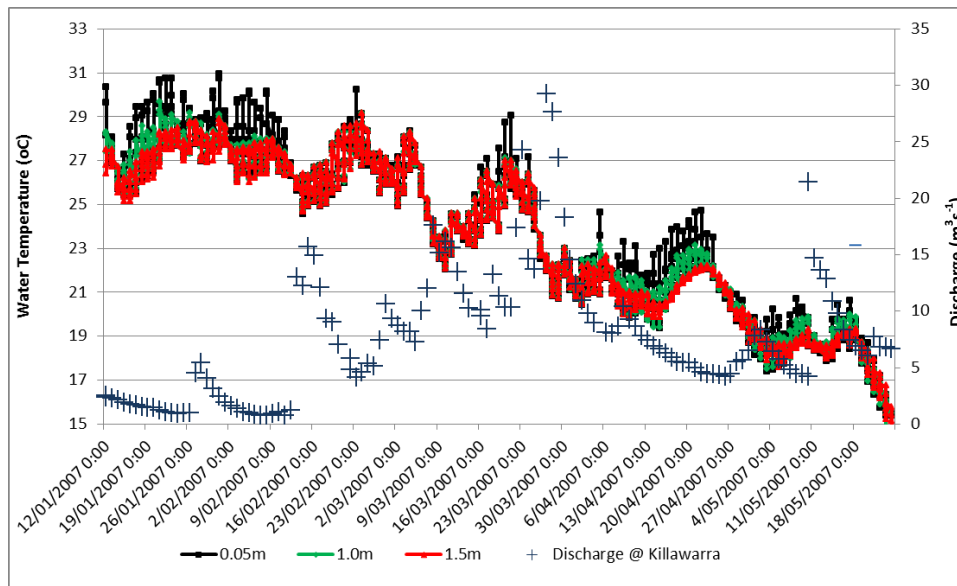


Figure 4.4 Water temperature in Railbridge pool macrophyte beds, January 2007 to May 2007

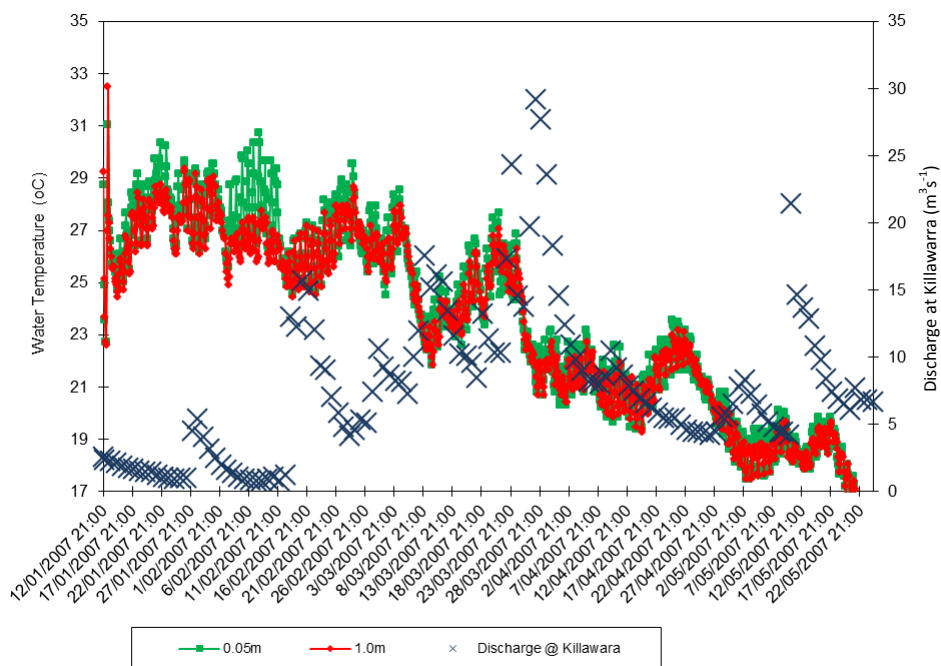


Figure 4.5 Water temperature in Woodside pool macrophyte beds, January 2007 to May 2007

Water temperature influences nearly all aspects of aquatic ecology (Allan 1995, Boulton and Brock 1999) and has been described as the most biologically important component of aquatic systems (Magnuson et al. 1979). Temperature variations within streams may be strongly influenced by the discharge of cold water releases from dams and suppressed flows increasing water temperatures (Mitrovic et al. 2003, Reinfelds and Williams 2012). Similarly to lakes, the lack of mixing in river pools, often from reduced discharge, can result in thermal stratification which drives chemical differences between the separated layers.

Another major temperature influence is on primary productivity, with increases in temperature generally resulting in increases in primary production (Tilley and Haushild 1975, O'Donohue and Dennison 1997). As discussed in Chapter 1, the shallow macrophyte-dominated pool areas tend to receive high sun-exposure and, as demonstrated by Figure 4.6, resulting in prolific growth of rooted aquatic plants over spring and summer (Thurtell and Bishop 2006). As water temperatures become excessively high during the summer low-flow periods, aquatic plant death and rapid decay often results in poor water quality (Thurtell and Bishop 2006). A biomass score calculated using the percentage cover of macrophytes and algae within shallow pools, and combined with summer heating potential (sun-exposed pool widths) highlighted that a shallow pool downstream of the confluence of the Manning and Gloucester Rivers was likely to be an area adversely affected under summer low flow conditions (Fig. 4.6) (Thurtell and Bishop 2006). There were also substantial differences in summer-heating potential between the shallow macrophyte-dominated pool known as the Railbridge Pool and the deep pool known as Ida Lake (Fig. 4.6).

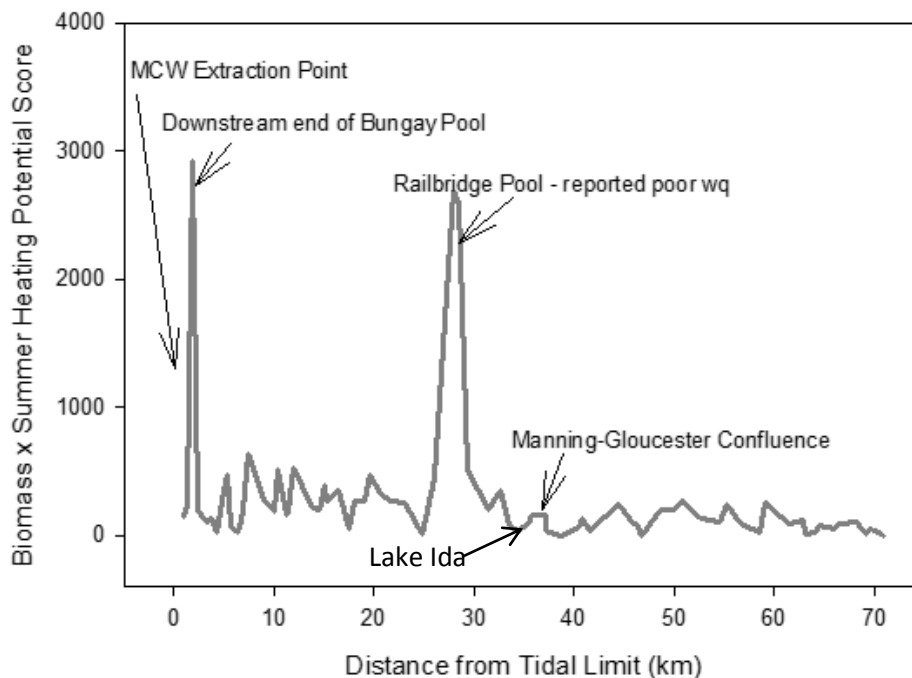


Figure 4.6 Macrophyte biomass and summer heating potential in shallow pools located in the Gloucester and Lower Manning Rivers, June/July 2006 (Source: Thurtell and Bishop 2006)

4.1.4 Aims and hypotheses

Nutrient cycling in aquatic ecosystems is governed by hydrologic processes, temperature and biological processes (Essington and Carpenter 2000). The aim of my study is to determine whether flow, water depth and temperature influence nutrient cycling in two deep pools in the Manning River. A further aim is to determine how interactions between primary productivity, flow and water temperature influence nutrient cycling within two shallow, macrophyte-dominated pools. This was undertaken through the measurement of nocturnal changes in water chemistry, coupled with the collection water temperature profile and discharge information over a longer temporal period.

I hypothesise that reduced flow (discharge and velocity), experienced under summer low flow conditions, will result thermal stratification and lower DO saturation at depth in two deep Manning River pools. I further predict that reduction in flow and low DO at depth within deep pools over the summer will increase nutrient concentrations in the water column through release from pool sediments.

I also aim to determine how interactions between primary productivity, summer flow and water temperature affect nutrient cycling in shallow-macrophyte-dominated pools. It is predicted that summer low flow conditions within shallow macrophyte-dominated pools will be correlated with low DO through increased oxygen consumption and reduced replenishment, possibly resulting in nocturnal anoxia. This reducing environment will result in increased nutrient concentrations within the water column through sediment nutrient release.

A further aim is to determine if summer low flow conditions will result in the exceedance of ANZECC and ARMCANZ (2000) and regionally-derived (developed in Chapter 3) water quality thresholds for the protection of aquatic ecosystems.

4.1.5 Study Site

During a survey of the Gloucester and lower Manning Rivers, pool attributes were measured and mapped (Thurtell and Bishop 2006) (Table 4.1). The existence of numerous lowland shallow (<3m), macrophyte-dominated pools was noted. The largest of these were named Railbridge, Woodside and Bungay (Table 4.1, Fig. 4.7). Two deeper pools (>8m) were located within the lower Manning River. One in the upstream section of Bungay Pool and the other was located downstream of the confluence of Gloucester River and is known as Ida Lake (Table 4.1, Fig. 4.7). All pools were dominated by sand substrates (Table 4.1).

Table 4.1 Manning River pool characteristics

Site	Sampling Depth (m)	Pool Width (m)	Thermistor Chain Depth (m)	Attributes
Ida Lake	16-18	190	1 x 18m	Deep pool located on a river bend; riffles located upstream and downstream; sandy substrate
Railbridge	0.5-1.5	95	1 x 1m 1 x 1.5m	Shallow, macrophyte dominated pool; riffle located downstream; sandy substrate
Woodside	0.5-1.5	65	1 x 1m 1x 1.5m	Shallow, macrophyte dominated pool; riffle located downstream; sandy/gravel substrate
Bungay	7-9	46	1 x 9m	Deep pool located on a river bend; riffle located upstream; sandy/cobbled substrate

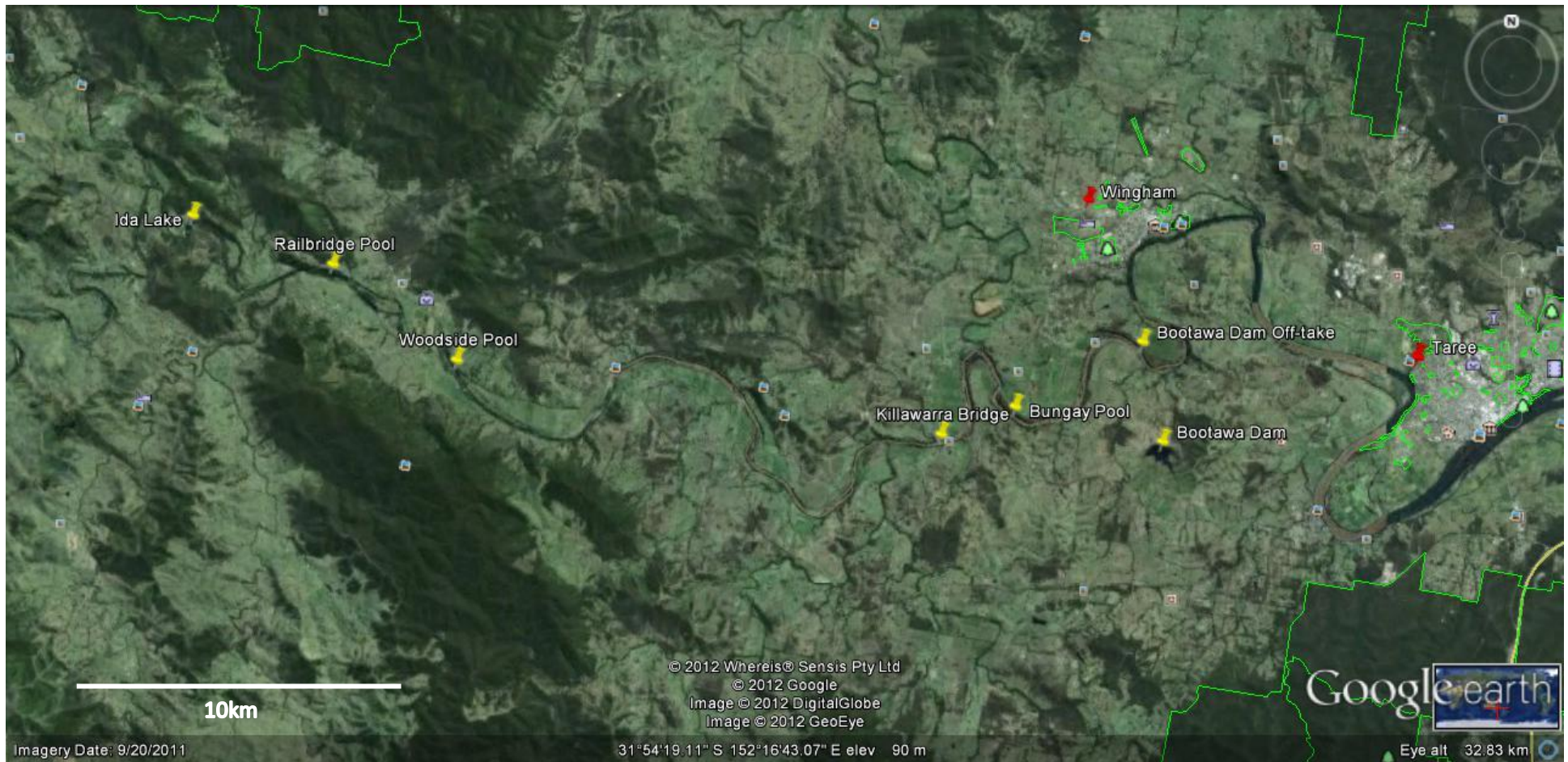


Figure 4.7 Manning River deep and shallow pool sites (Source: Google Maps. 2011)

Railbridge Pool and Ida Lake were selected to investigate nocturnal changes in nutrients. These pools were identified as representative examples of these types of systems within the lower Manning River and had previously been identified as possibly influencing nutrient type and availability, with impacts on downstream environments (Thurtell and Bishop 2006).

Ida Lake, the deepest pool identified in the lower Manning River, is approximately 17-18 m in depth. It is a large pool, approximately 190 m wide and 250 m long (Fig. 4.8). Riffles are situated upstream and downstream of the deep pool area, with steep bank running along the left side of the pool (looking downstream) (Fig. 4.9). Railbridge Pool is located a short distance downstream of Ida Lake and slightly upstream of Mount George. It is a long shallow pool with an average depth of approximately 1.5 to 2m (Fig. 4.10). A long riffle is located downstream of the pool area, with the left side of the pool (looking downstream) flanking the pool (Fig. 4.10). There is anecdotal evidence of poor water quality under high temperatures and low flow conditions due to prolific plant and algal growth (Fig. 4.11).

Dominant macrophytes observed within both the shallow pool sites were predominantly submerged, attached and included *Myriophyllum sp.*, *Potamogeton, sp.* *Vallisneria sp.* and *Hydrilla sp.* These plants made up very thick mats throughout the pool and supported large amounts of epiphytic algae as illustrated in Fig. 4.12.



Figure 4.8 Ida Lake buoy marking deepest part of the pool with temperature data loggers attached.



Figure 4.9 Aerial view of Ida Lake (source: Google Earth, accessed 25 November 2013)



Figure 4.10 Aerial view of Rail Bridge Pool (source: Google Earth, accessed 25 November 2013)



Figure 4.11 Macrophyte growth in Railbridge Pool, November 2008



Figure 4.12 Prolific aquatic plant and epiphytic growth in lower Manning River, August 2007

4.2 Methods

4.2.1 Field sampling

Nocturnal trial

To determine nocturnal influences of flow, water depth and temperature on nutrient cycling in deep pools in the Manning River, in situ water quality data and nutrient samples were collected from vertical profiles at the deepest section of Ida Lake, at 16-18 m (epibenthic), 11-12-m, 3-4 m and 0.05 m (near surface) every 3 hours for 12-hours, commencing at 6pm 19th January and finishing at 6am on the 20th January 2009. To assess nocturnal influences on interactions between primary productivity, flow and water temperature influence nutrient cycling within shallow, macrophyte-dominated pools, the pool known as Railbridge Pool was sampled for 12-hours commencing at 6pm on 4th February 2009, samples were collected from the surface and epibenthic zones of macrophyte beds situated in 1.5 m and 1 m water depth. Surface and epibenthic (depth 1.5 m) samples were also collected at the same time from an open water section.

Flows in the Manning River at Killawarra gauging station, located downstream of both Ida Lake and Railbridge Pool (see Fig. 4.7), were reduced ($<20\text{m}^3\text{s}^{-1}$) for at least 4 weeks leading up to the nocturnal study (Fig. 4.13). As these flow conditions were not repeated during the study period the nocturnal assessment could not be repeated for inclusion in this thesis.

Nutrient release trial

To determine the timing and duration of stratification or anoxic conditions, thermistor chains deploying Hobo temperature loggers were used to record temperature every 3 hours. Table 4.1 indicates the depth to which loggers were deployed at each site. Logger information was collected from January 2007 to January 2009 at Ida Lake and Railbridge Pool, along with two other pool sites, Bungay Pool (a deep pool downstream of Ida lake and Railbridge Pool) and Woodside Pool (shallow, macrophyte-dominated, upstream of Bungay Pool but downstream of Ida Lake and Railbridge Pool) (Table 4.1, Fig. 4.7). The data loggers were used to measure the preceding and prevailing conditions within Manning River pools.

A Hydrolab DS 5X multiprobe was used to undertake profile measurements of dissolved oxygen (DO), pH, electrical conductivity and oxidation reduction potential (ORP) at all sites as part of this study. Nutrients - total phosphorus (TP), soluble reactive phosphorus (SRP), total nitrogen (TN), ammonium (NH_4^+), nitrite (NO_2) and nitrate (NO_3) - were collected at 0.5 to 1m intervals within Ida Lake, Railbridge Pool, Bungay Pool and Woodside Pool on a monthly basis.

Water quality variables including dissolved oxygen (DO - % saturation and mgL^{-1}), pH, electrical conductivity (EC - $\mu\text{S cm}^{-1}$) and water temperature were measured through the profiles of both pools using a Hydrolab (Table 4.2). Samples were also collected for TN, NO_3 , NO_2 , NH_4^+ , TP and SRP. Vertical profiles of water quality and nutrients were taken every 1 m within the deep pools (see Table 4.1) and every 0.5 m from the shallow pools using a small pump and hose to suction water from the various depths. Samples were chilled immediately following collection, where necessary filtered and frozen on return from the sampling trip. The samples were analysed by MCW Laboratory located at Bootawa Dam.

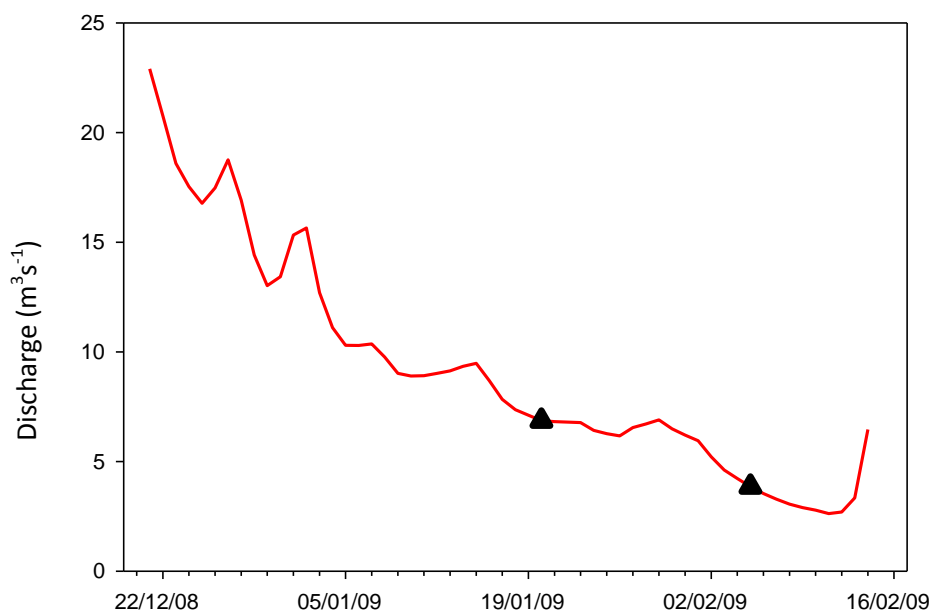


Figure 4.13 Discharge in the Manning River at Killawarra during the experiments and sampling times, December to February 2009 (Ida Lake sampling = 20/1/09; Railbridge Pool sampling = 4/2/09)

4.2.2 Laboratory analyses

MCW Laboratory is accredited by NATA (12902) for technical competence in Chemical (14262) and Biological (12895) testing in accordance with ISO/IEC 17025. Nutrient samples were thawed prior to colorimetric analysis in the laboratory. TP and TN samples were digested with persulfate (Hosomi & Sudo 1986), and the concentrations analysed using a spectrophotometer. SRP was analysed using the molybdate-blue method (Murphy and Riley 1962) and $\text{NH}_{3/4}\text{-N}$ and $\text{NO}_3\text{-N}$ was analysed using cadmium-copper reduction (Wood et al. 1967). Refer to Section 3.3 for further details on water quality analyses.

Table 4.2 Details of Hydrolab multiprobe and field turbidity meter

Sensor	Range	Accuracy	Resolution
Hach LDO	0-60 mg L ⁻¹	± 0.1 mg L ⁻¹ at <8 mg L ⁻¹ ± 0.2 mg L ⁻¹ at >8 mg L ⁻¹	0.01 mg L ⁻¹
Hach ORP	-1500 to 1500 mV	± 0.1 mV	0.1 mV
Temperature (30k ohm thermistor)	-5 to 50 °C	± 0.10°C	0.01°C
Conductivity (graphite electrodes)	0-100 mS cm ⁻¹	± 0.5% reading + 0.001 mS cm ⁻¹	0.001 mS cm ⁻¹
pH (reference electrode)	0 to 14 pH units	± 0.2 units	0.01 units

4.2.3 Data analyses

Nocturnal trial

Nocturnal profiles at depth over time were used to determine the strength of stratification and influences on water quality within the study area. Accompanying these data was historical data supplied by MCW which compared diurnal TP concentrations to water level at the Bootawa Dam offtake, located downstream Bungay Pool.

Principal components analysis (PCA) was used to investigate relationships between nutrient and environmental variables to sites. Values for temperature, depth of water, DO% saturation, pH, TN, NH_4^+ , NO_x , TP and SRP were used to investigate relationships for nocturnal changes at Ida Lake and Railbridge pool. All values except for pH, temperature and water depth were log-transformed to improve normality and normalised to a common measurement scale to facilitate comparison using Euclidean distance (Clarke and Warwick 2001). Multivariate statistical analyses were done using PRIMER Version 6 (PRIMER-E Ltd, Plymouth).

To test how changes in nutrient concentrations under low flows may influence biotic responses, molar ratios for nutrients were calculated for TN:TP, TN: NO_x and TP:SRP using nocturnal profile data collected from Ida Lake and Railbridge Pool. Changes relating to depth and stratification and how these changes may impact on primary production were also measured.

Nutrient release trial

As it was predicted that nutrients would exceed thresholds for the protection of aquatic ecosystems, all nutrient data were compared to the Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZECC and ARMCANZ 2000) and regionally-derived thresholds developed in Chapter 3 of this thesis. These guidelines provide a basis for setting water quality objectives to sustain environmental values for this water resource. The guidelines also provide threshold values for a chemical or nutrient that, if exceeded, have the potential to cause a problem and, therefore, should result in a management response.

Discharge was graphically compared to physical and chemical variables to determine the relationship of flow to the presence and persistence of stratification and to determine the influence of stratification on water quality.

4.3 Results

4.3.1 Effects of water temperature and flow on stratification

4.3.1.1 Deep pools

Stratification was detected in both deep pools in the summer of 2008/09 under low flow conditions, and was particularly strong in Ida Lake (Fig. 4.14). During this stratification period differences in DO and water temperature were recorded within both deep pools (Figs. 4.15 and 4.16). Discharge at this time in the Manning River was below $7.5 \text{ m}^3\text{s}^{-1}$ at Killawarra.

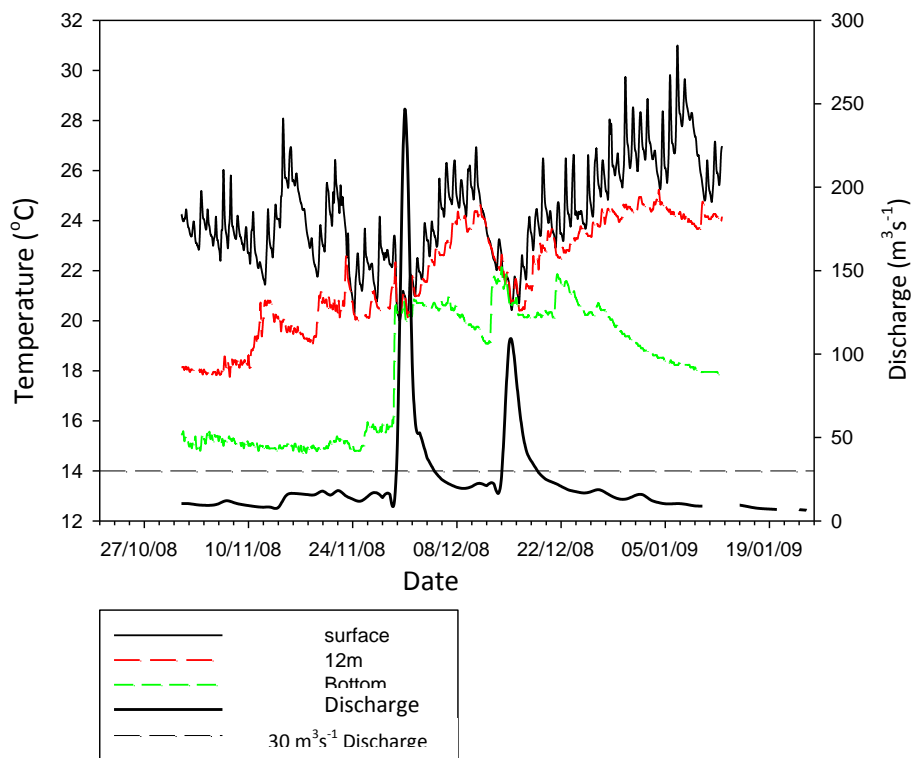


Figure 4.14 Ida Lake water temperature profile surface, 12m depth and bottom, November 2008 to January 2009

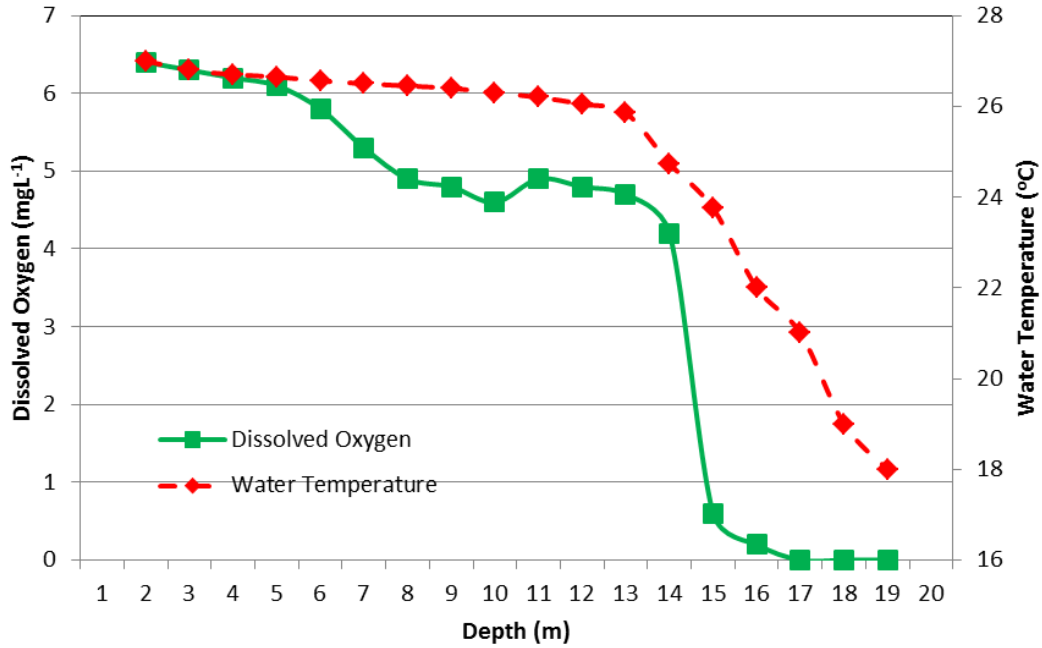


Figure 4.15 Dissolved oxygen and water temperature at various depths in Ida Lake, Manning River, 2pm 20th January 2009

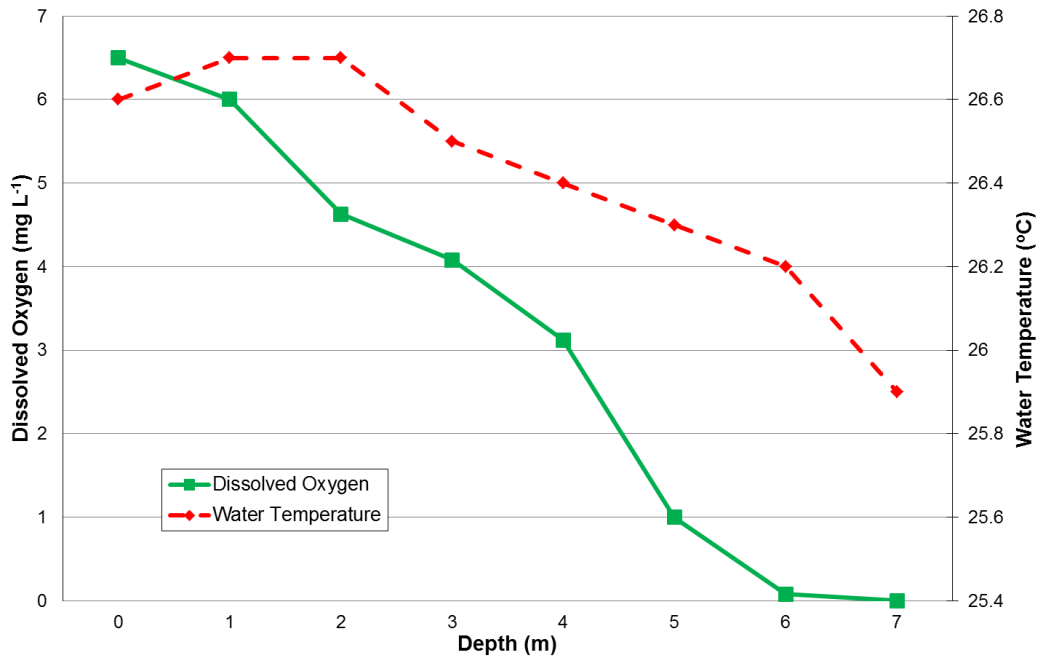


Figure 4.16 Dissolved oxygen and water temperature at various depths in Bungay Pool, Manning River, 2pm 6th February 2009

To further determine the influence of flow and temperature on the deepest of these pools, Ida Lake, a nocturnal study was undertaken to see if physical or chemical changes occurred overnight. Figure 4.17 indicates that large water temperature differences occurred between the bottom and surface layers of Ida Lake during the overnight sampling period. The maximum temperature difference between the surface and bottom waters was approximately 9°C, with surface

temperatures around 27.5°C and 18.5°C recorded within the hypolimnion. Discharge at this time was 7.1 m³s⁻¹ at Killawarra gauging station. The hypolimnion appeared to commence below 12-13 m from the water surface, with the metalimnion extending over approximately 2m, from 12m to 14m below the pool surface (Fig. 4.17).

Measurement of water temperature profiles in Lake Ida revealed little change through the night, with a decline of 1°C at the water surface as exchange occurred between the cooler night air and a convergence in temperature in the top 5m (Fig. 4.17).

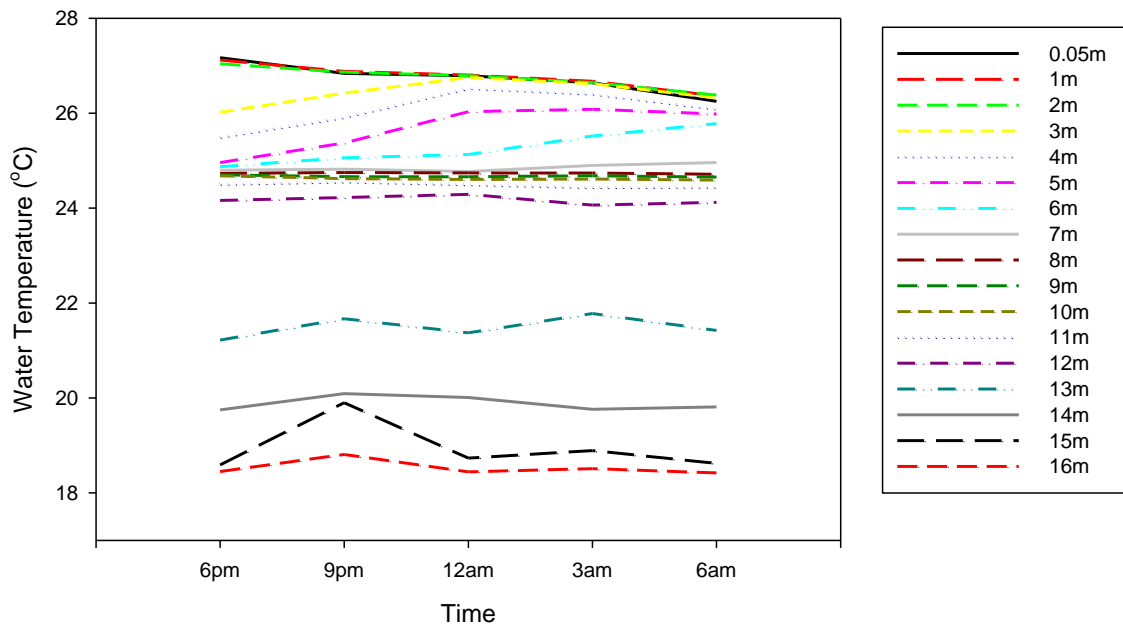


Figure 4.17 Nocturnal water temperature profiles, Ida Lake, 20th January 2009

4.3.1.2 Shallow Pools

To determine the influence of short-term stratification on nutrients in shallow macrophyte-dominated pools, three sites within the Railbridge pool were measured throughout the night. Two sites contained dense macrophyte growth, one being 1.5 m deep and the other 1 m deep, and a third area being 1.5 m deep and containing no macrophytes. Water temperature at all sites indicated a small, gradual decline during the night. The greatest decline was measured within the 1.5 m open water site, with water temperatures dropping 1.3°C over the 12-hour period (Fig. 4.18). Differences between surface and bottom layers at all three sites were minimal (Fig. 4.18).

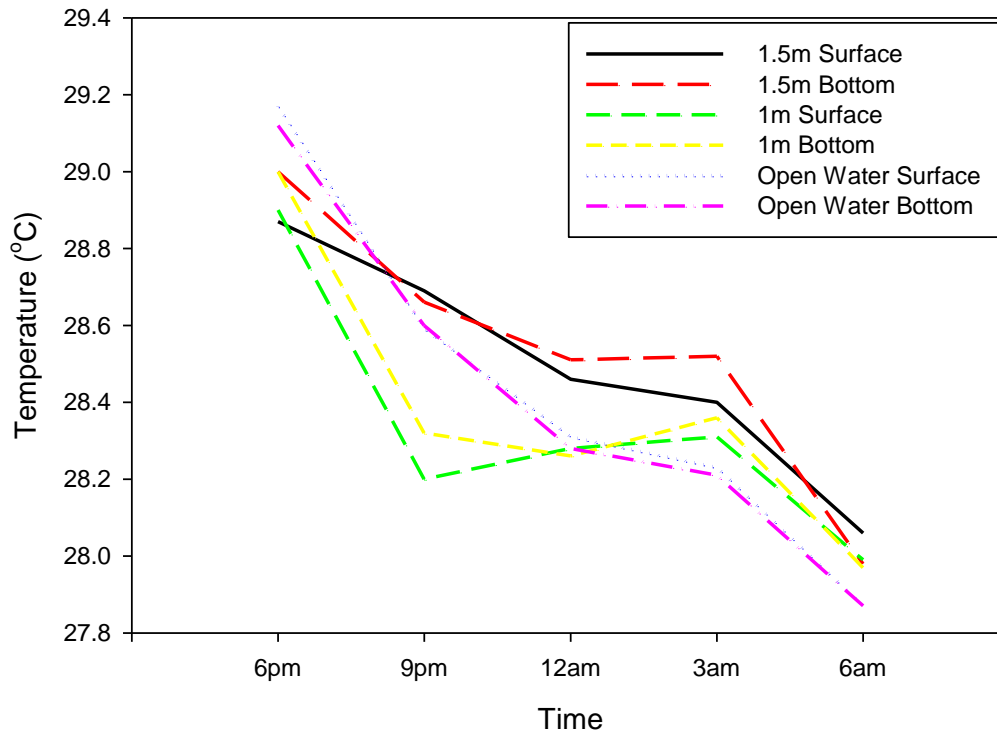


Figure 4.18 Nocturnal water temperature profiles, Railbridge Pool, 4th February 2009

4.3.2 Stratification persistence

4.3.2.1 Deep pools

Routine sampling undertaken during the Austral summer of 2009 indicated that thermal stratification reduced oxygen saturation to near 0 mgL^{-1} in the hypolimnion of Ida Lake and Bungay pools, resulting in hypoxic conditions (Figs. 4.19 and 4.20). Discharge at Killawarra during this January period ranged from 6 to $16 \text{ m}^3\text{s}^{-1}$, well below the $30 \text{ m}^3\text{s}^{-1}$ required to breakdown thermal stratification in these pools. These low flows continued until 15 February 2009, when flows exceeding $500 \text{ m}^3\text{s}^{-1}$ were recorded at the Killawarra gauge. These flows resulted in the destratification of both deep pools (Fig. 4.21). As water temperature increased during spring 2009 and flows reduced at Killawarra to $8 \text{ m}^3\text{s}^{-1}$ thermal stratification began to reform in both pools (Figs. 4.22 and 4.23).

The nocturnal study conducted within Ida Lake in 2009 supported the long-term data (Fig. 4.24). The 12-hour study indicated that, while there was slight drop in surface water dissolved oxygen saturation during the night, most likely due to planktonic respiration, the hypoxic conditions were maintained below a depth of 12 m (Fig. 4.24).

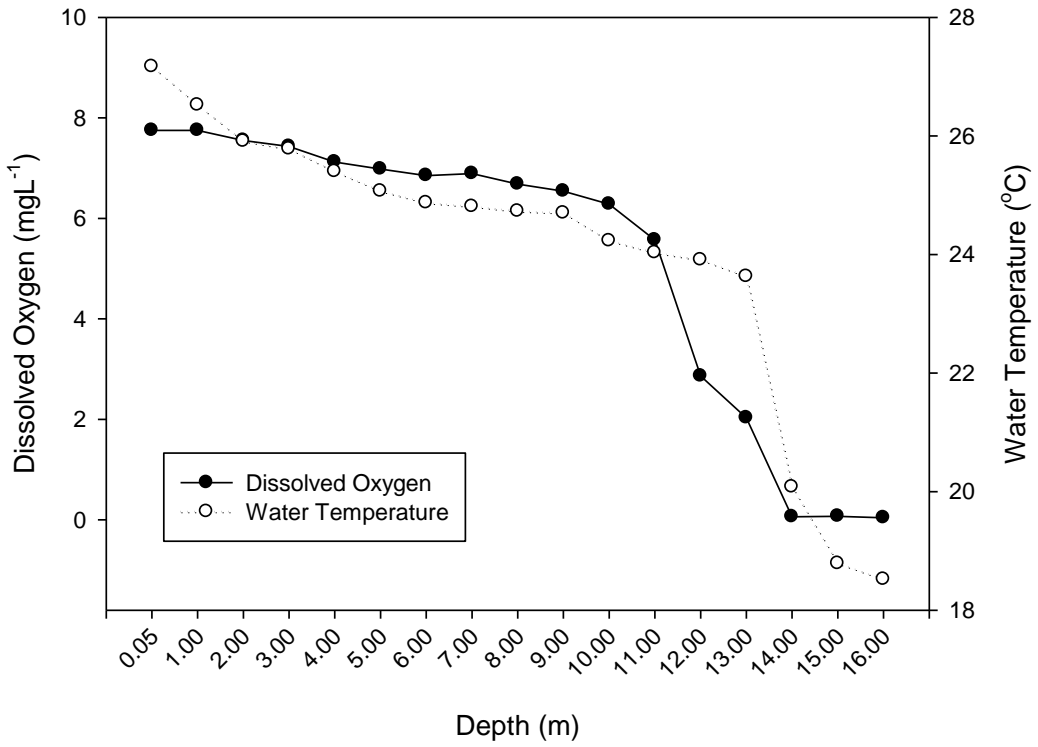


Figure 4.19 Water temperature and dissolved oxygen profiles, Ida Lake, 3pm 12 January 2009

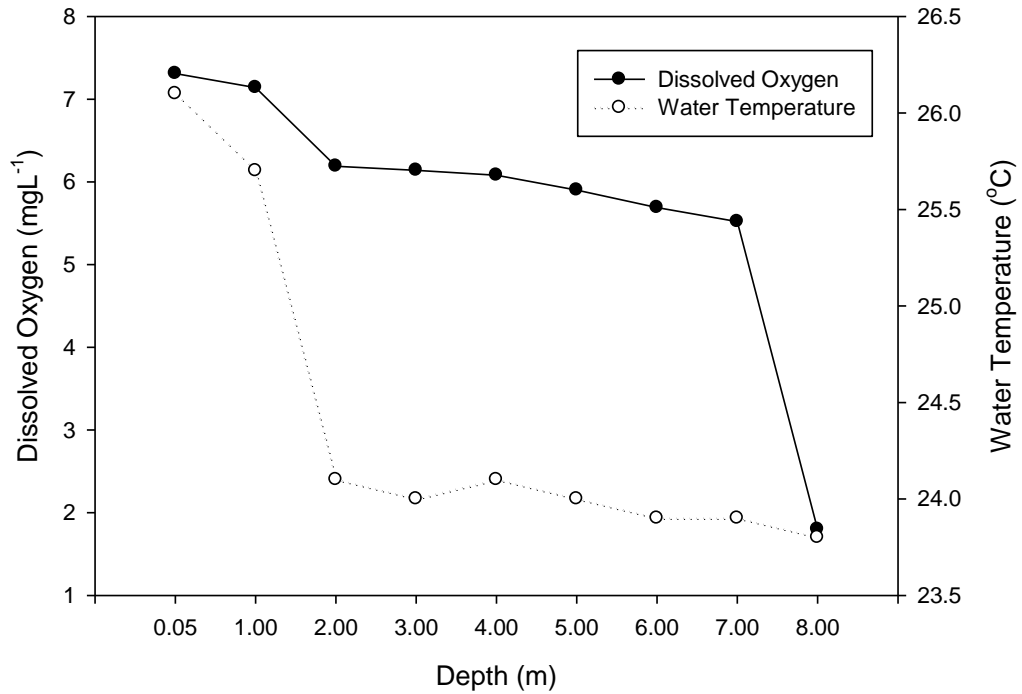


Figure 4.20 Water temperature and dissolved oxygen profiles, Bungay Pool, 11am 10 January 2009

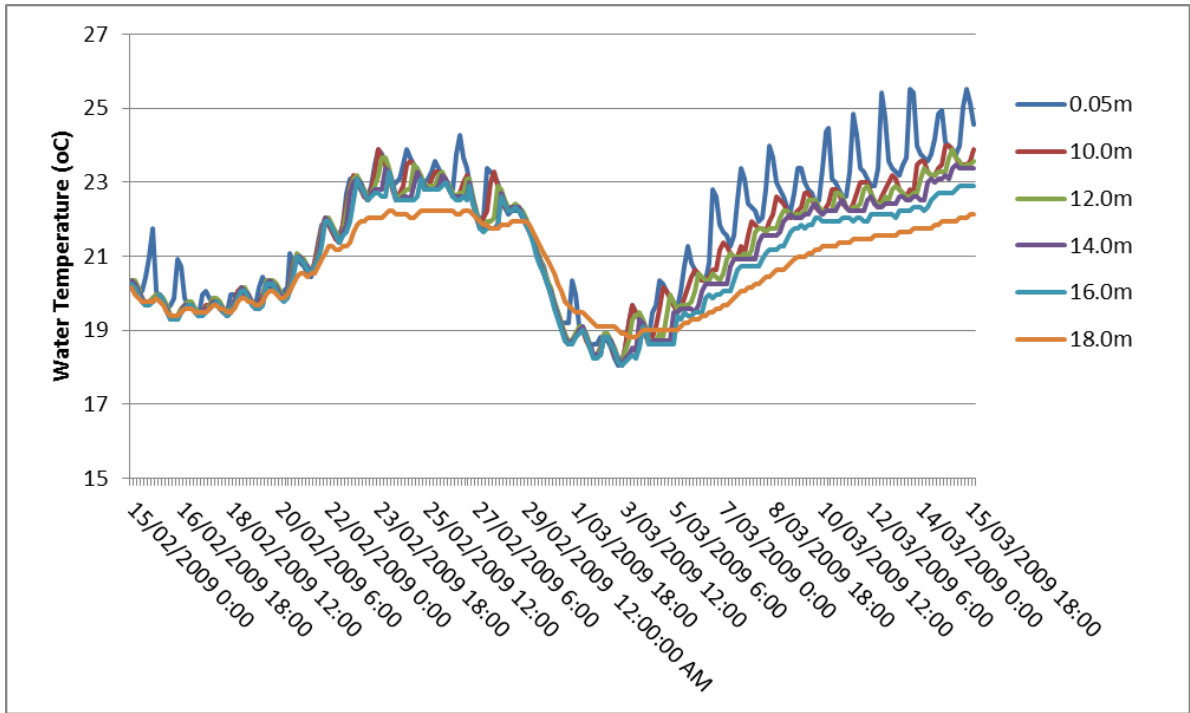


Figure 4.21 Water temperature profile, Ida Lake, February to March 2009.

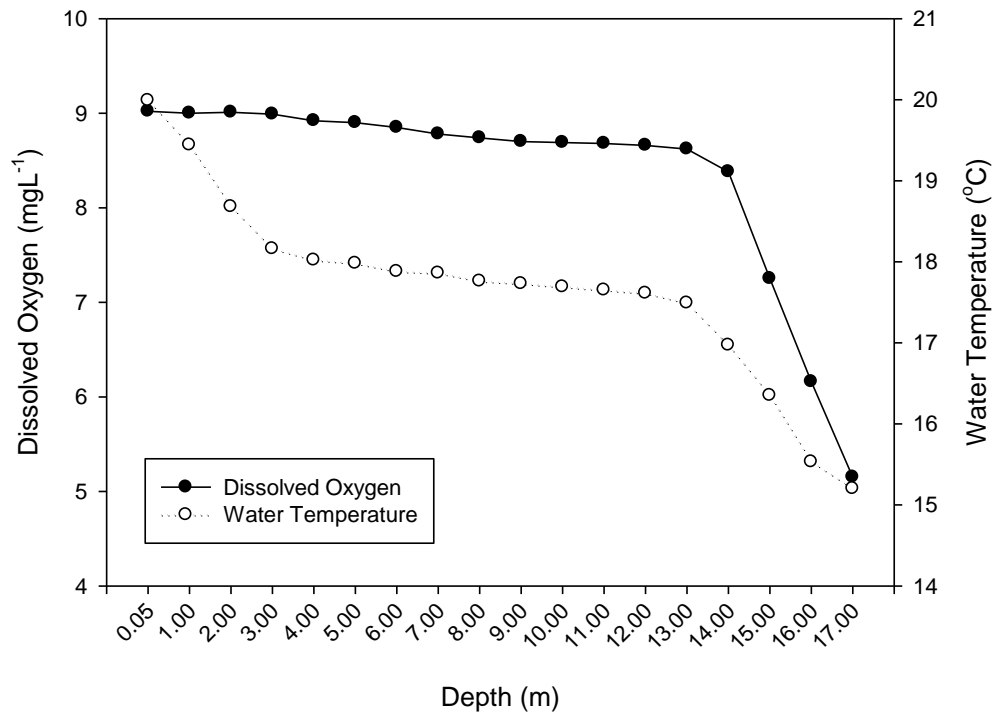


Figure 4.22 Water temperature and dissolved oxygen profiles, Ida Lake, 1pm 2 October 2009

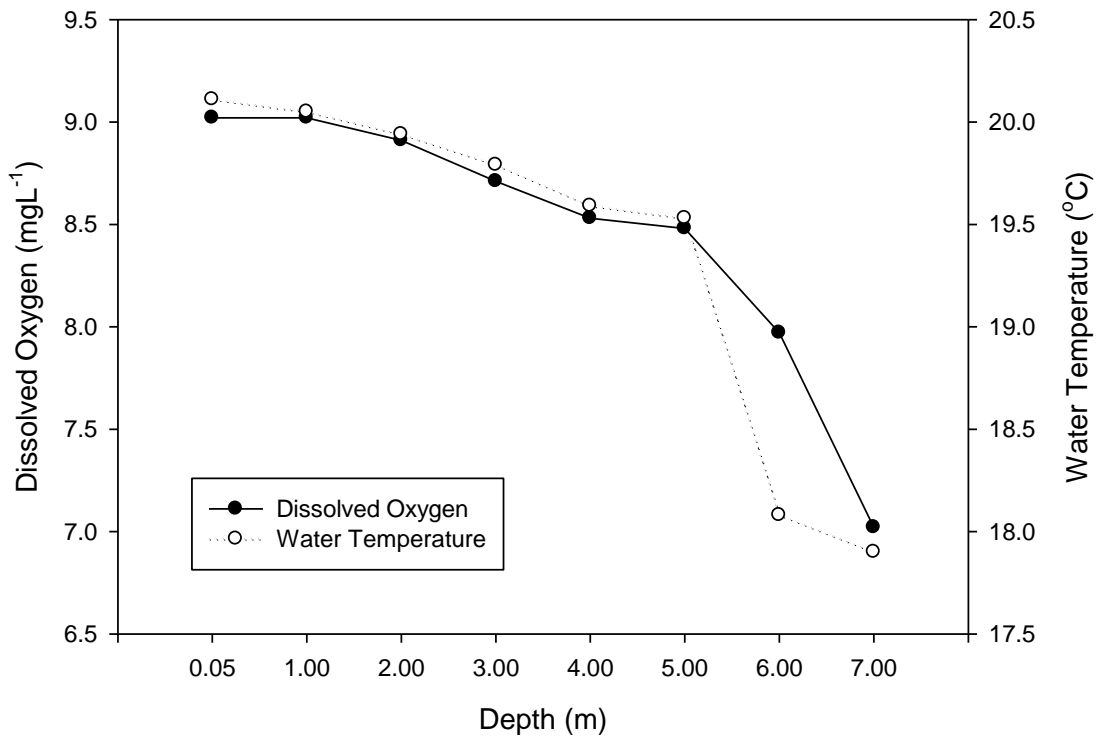


Figure 4.23 Water temperature and dissolved oxygen profiles, Bungay Pool, 12pm 3 October 2009

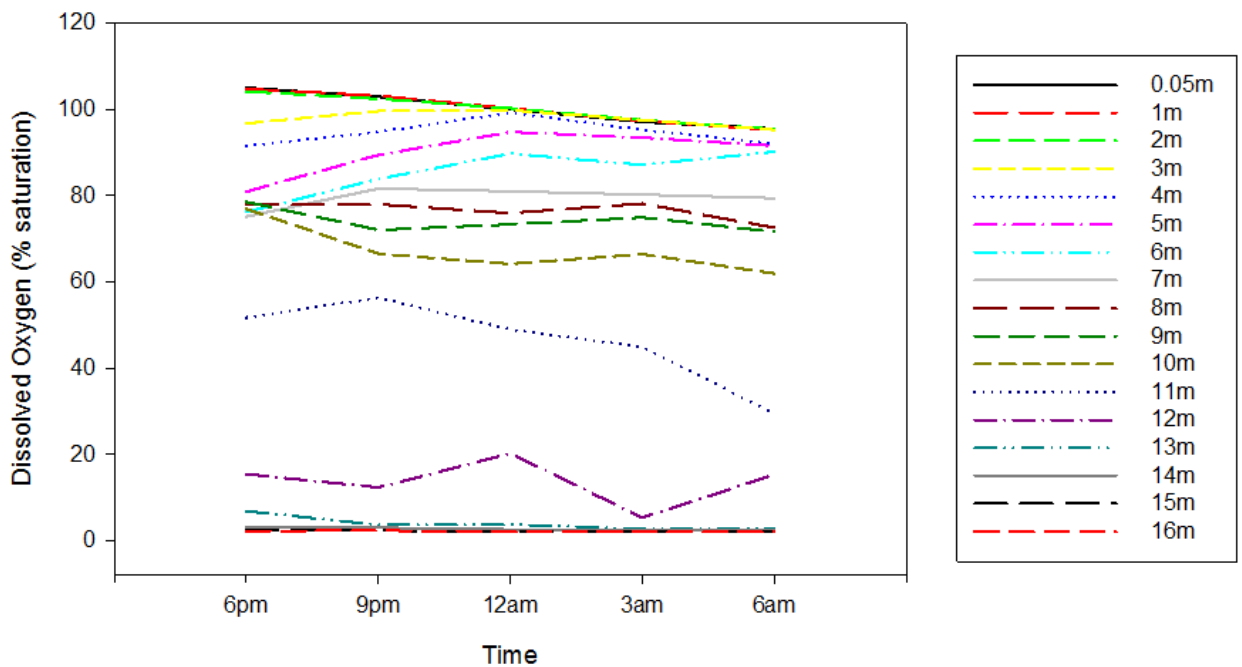


Figure 4.24 Nocturnal dissolved oxygen saturation profiles, Ida Lake, 20th January 2009

The top 2 m of Ida Lake exhibited pH levels of 7.9 (Fig. 4.25). Interestingly, an increase of 0.3 pH units was measured in the 3-7 m range which may indicate a change in profile position of the phytoplankton or diel convection of surface waters (Fig. 4.25). However, a difference 1.2 pH units remained between the surface and bottom layers within Ida Lake throughout the 12-hour sampling period (Fig. 4.25). A profile measurement conducted within Bungay Pool in February 2009 shows a similar pattern of declining pH with depth (Fig. 4.26).

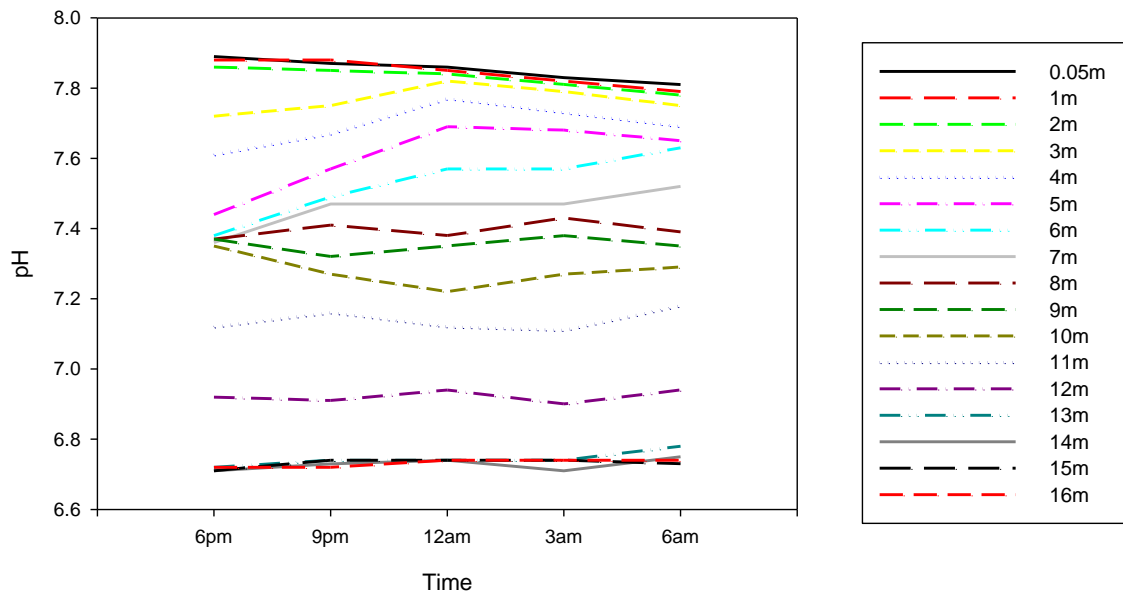


Figure 4.25 Nocturnal pH profiles, Ida Lake, 20th January 2009

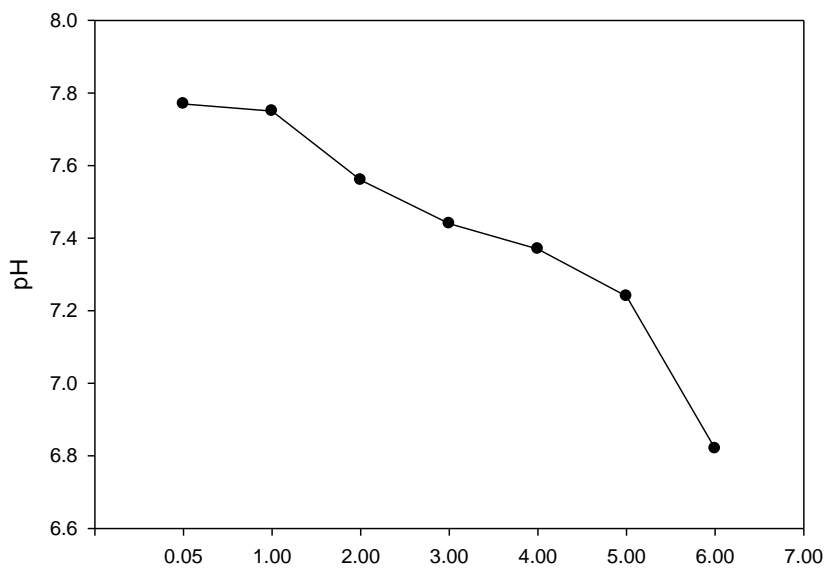


Figure 4.26 pH profile, Bungay Pool, 2pm 6th February 2009

Electrical conductivity increased by $140 \mu\text{S cm}^{-1}$ from the epilimnion to the hypolimnion of Ida Lake (Fig. 4.27). Despite the large difference between surface and bottom layers, electrical conductivity remains within the 80th percentile of the regionally-derived threshold of $550 \mu\text{S/cm}$ (see Chapter 3). A similar pattern of increasing conductivity with depth under low flows was found within Bungay Pool in February 2009 (Fig. 4.28).

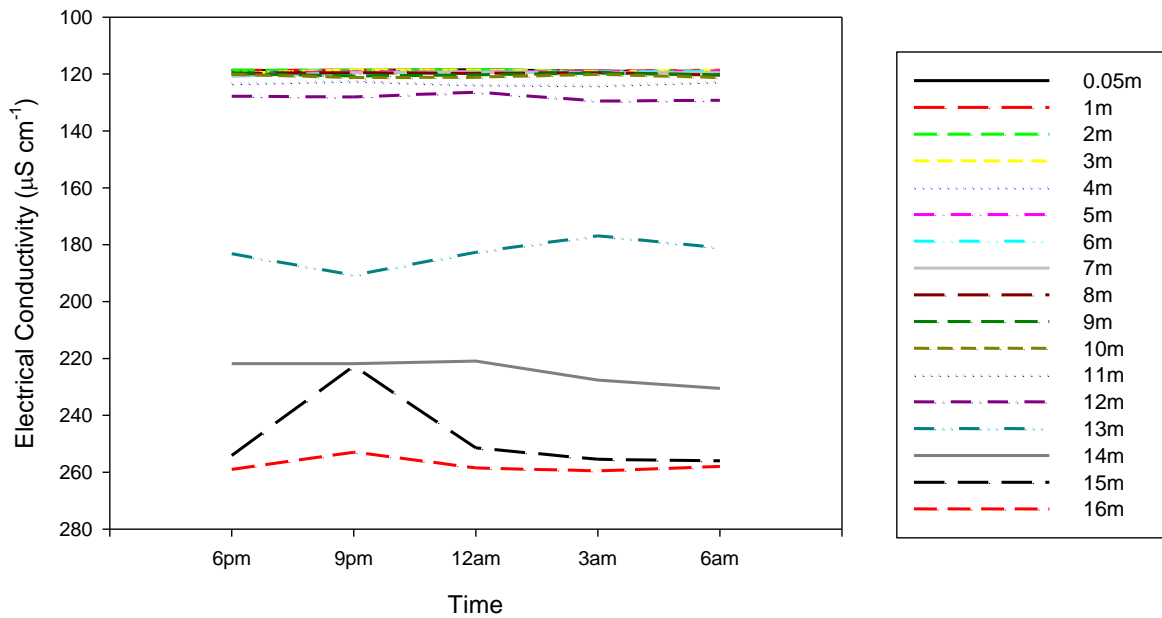


Figure 4.27 Nocturnal electrical conductivity profiles, Ida Lake, 20th January 2009

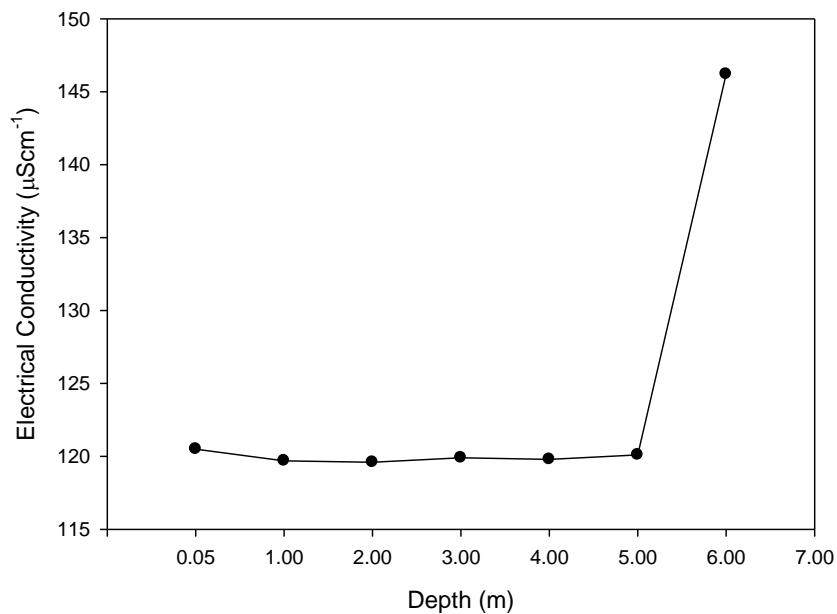


Figure 4.28 Electrical conductivity profile, Bungay Pool, 2pm 6th February 2009

Oxidising reduction potential (ORP), measured during December and January 2009, indicated a substantial decline through the profile (Fig. 4.29). All ORP values throughout the shallow-macrophyte dominated pools were above 350 mV, indicating an oxidising environment.

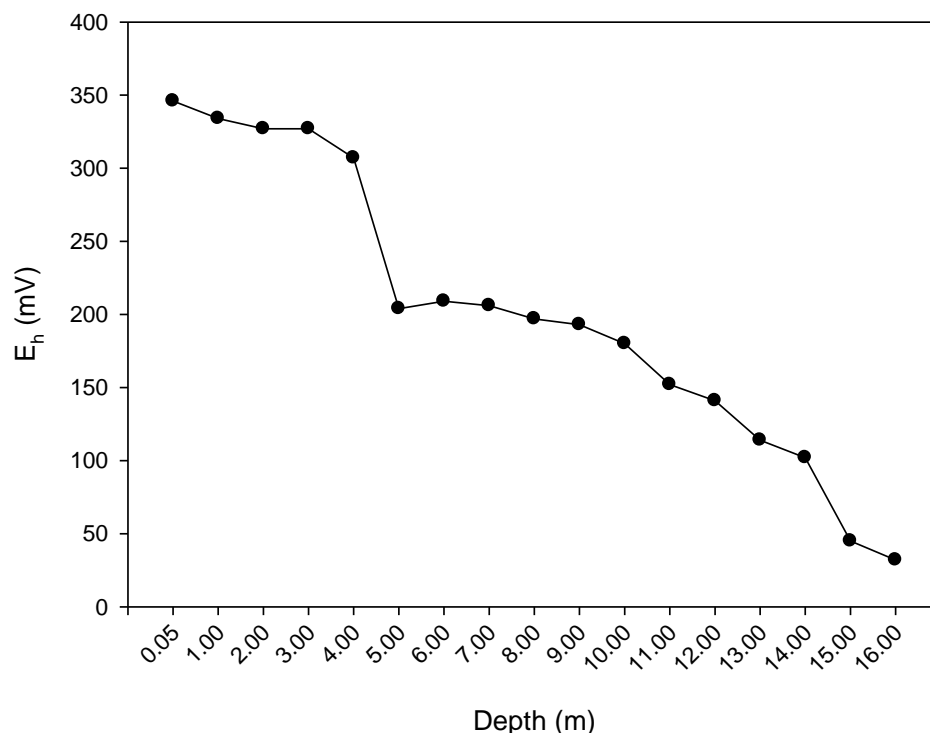


Figure 4.29 Redox potential within Ida Lake, 3pm 12th January 2009

Results from the nocturnal study undertaken in 2009 showed that nutrient concentrations fluctuated around the 80th percentile threshold in the epilimnion, while concentrations in the hypolimnion were approximately 5 times greater than the recommended threshold throughout the 12-hour period (Fig. 4.30). Ammonium concentrations also greatly exceeded the 80th percentile threshold of 0.02 mg/L within the hypolimnion, and at times, within the metalimnion (Fig. 4.31).

Concentrations of TN and ammonium in the hypolimnion dropped by 0.5 mgL⁻¹ between 6pm and 9pm within Ida Lake (Figs. 4.30 and 4.31). From this reduced concentration, nitrogen appears to gradually increase (Figs. 4.30 and 4.31). There is an increase in surface nitrogen and ammonium between 3am and 6am (Figs. 4.30 and 4.31). This may be influenced by an accompanying small decrease in water temperature, dissolved oxygen and pH in the epilimnion (see Figs. 4.22, 4.24 and 4.25).

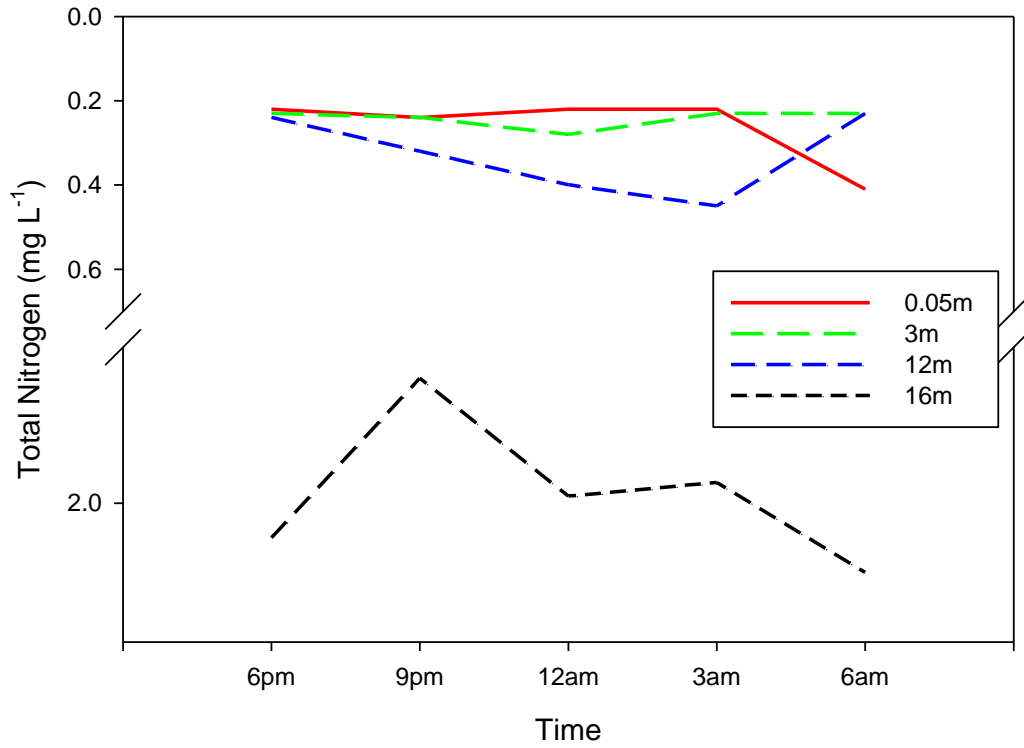


Figure 4.30 Nocturnal total nitrogen profiles, Ida Lake, 20th January 2009
 (Y axis is inverse with highest concentrations at the bottom of the graph)

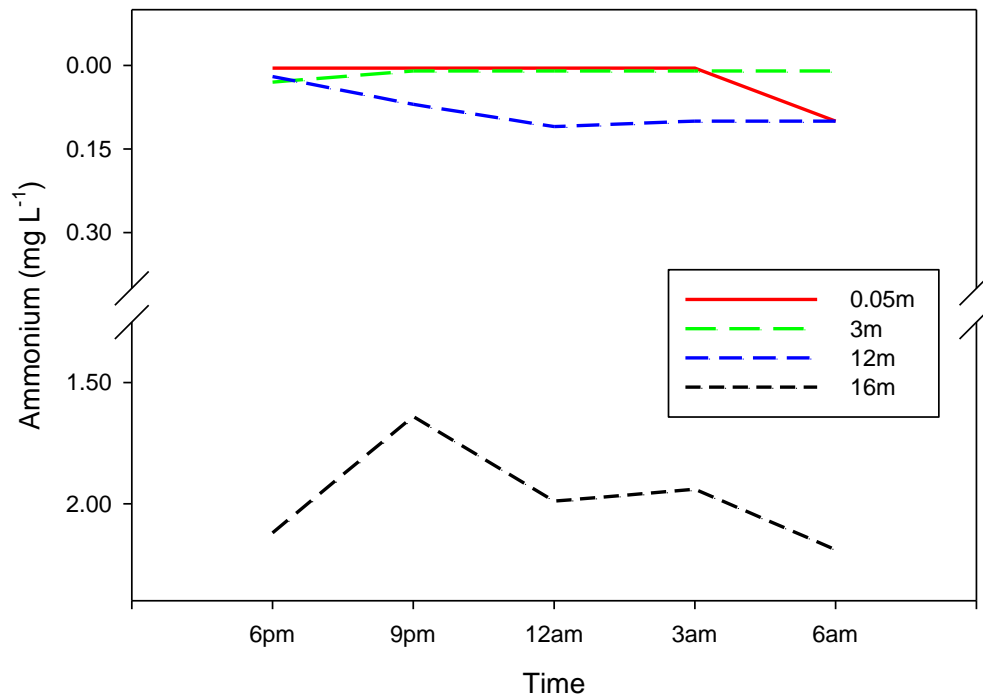


Figure 4.31 Nocturnal ammonium profiles, Ida Lake, 20th January 2009
 (Y axis is inverse with highest concentrations at the bottom of the graph)

Results from the Ida Lake nocturnal study undertaken in 2009 showed that TP concentrations fluctuated around the 80th percentile threshold for regionally-derived thresholds in the epilimnion, with an increase above the threshold identified at 3 m and 12m depths at 3am, while concentrations in the hypolimnion were approximately 30 times greater than the recommended threshold throughout most of the 12-hour period (Fig. 4.32). This pattern in TP concentrations is similar to TN and ammonium, with a decrease measured between 6pm and 9pm (Fig. 4.32).

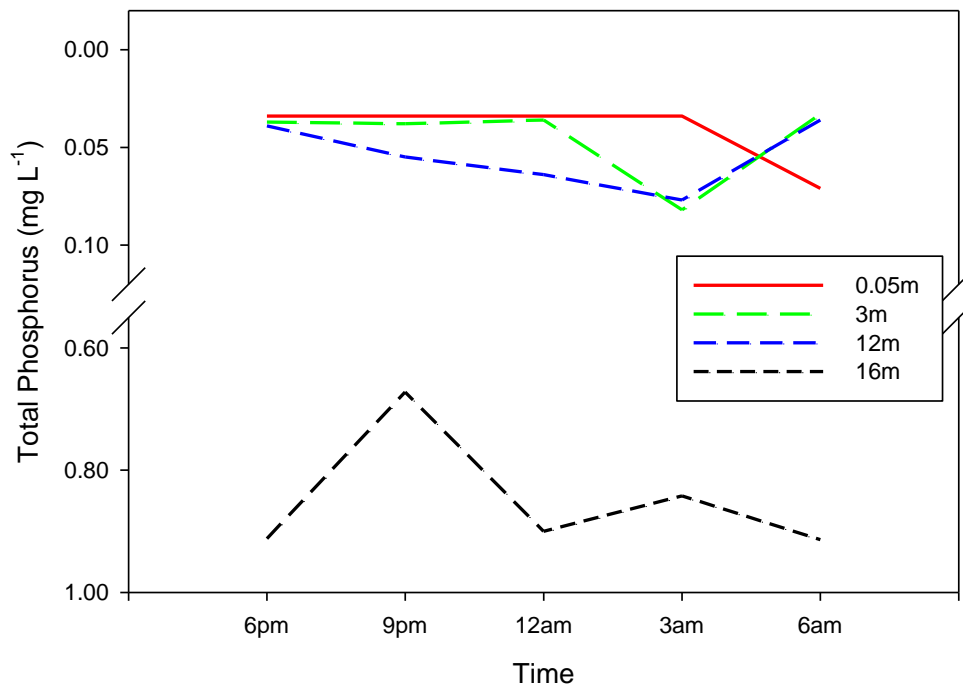


Figure 4.32 Nocturnal total phosphorus profiles, Ida Lake, 20th January 2009

(Y axis is inverse with highest concentrations at the bottom of the graph)

Soluble reactive phosphorus was generally within the 80th percentile regionally-derived threshold of 0.02 mgL⁻¹ during the study period, slightly exceeding it on occasion within the thermocline (Fig. 4.33). The exceedance of SRP in the thermocline demonstrates a different pattern from the other measured nutrients in Ida Lake. Concentrations are lowest within the hypolimnion and highest at the 12-m thermocline-hypolimnion interface throughout the 12-hour period (Fig. 4.33). Concentrations then increased within the surface layer between 3am and 6am, as was also found with TP (Fig. 4.33).

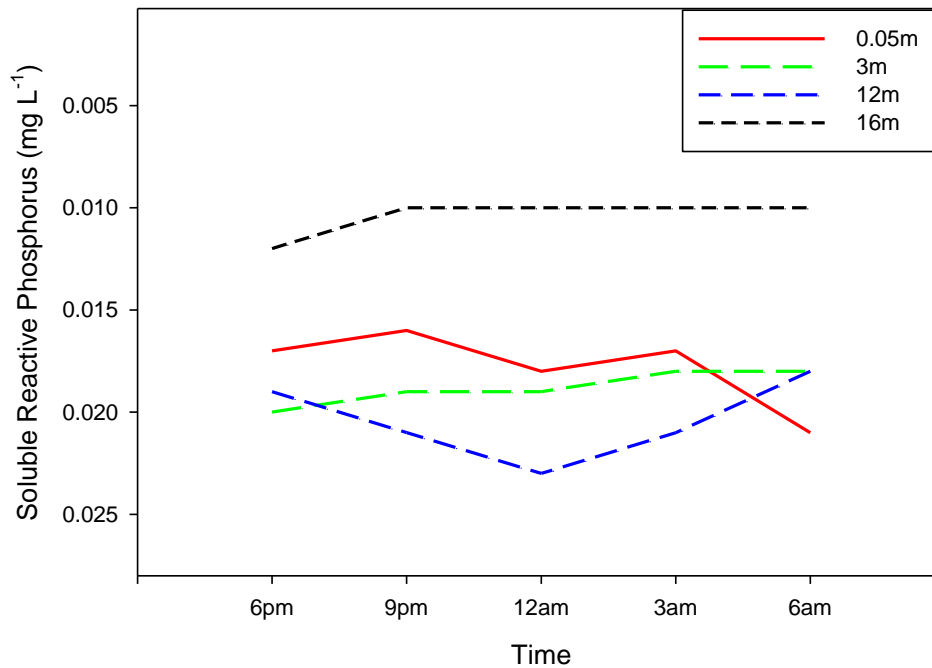


Figure 4.33 Nocturnal soluble reactive phosphorus profiles, Ida Lake, 20th January 2009
 (Y axis is inverse with highest concentrations at the bottom of the graph)

Nutrient ratios, calculated for TN:TP, TN:NO_x and TP:SRP indicated that the epibenthic zone generally supported very different ratios than other zones within Ida Lake (Fig. 4.34). Within this zone, TN:TP ratios remained around 2:1, with ratios at other depths generally between 6:1 and 8:1 (Fig 4.34). The exception to this was found at 4 m below the water surface at 3:30am which presented a ratio of 2:1, when TP almost doubled (Figs. 4.32 and 4.34). Ratios less than 16:1 have been considered TN limiting for periphyton, if this is the case it would only relate to available N as TN concentrations within the epibenthic zone reached 2.2 mgL⁻¹ during this study.

TN:NO_x ratios demonstrated large fluctuations within the epibenthic zone, which had ratios of up to 10 times greater than ratios measured in the other zones (Fig. 4.35). Ratios ranged from 164:1 to 366:1 in the epibenthic zone, while ratios ranged from 20:1 to 80:1 in the other zones (Fig. 4.35). TN:NO_x ratios are indicative the possible lack of available N in comparison to TN.

TP:SRP ratios were extremely high in the epibenthic zone, reaching 90:1 in comparison to the other zones which were generally around 3:1 (Fig. 4.36). This large difference in ratios between the epibenthic zone and the rest of the water column is driven by the exceptionally high TP concentrations of between 0.67 to 0.91 mgL⁻¹.

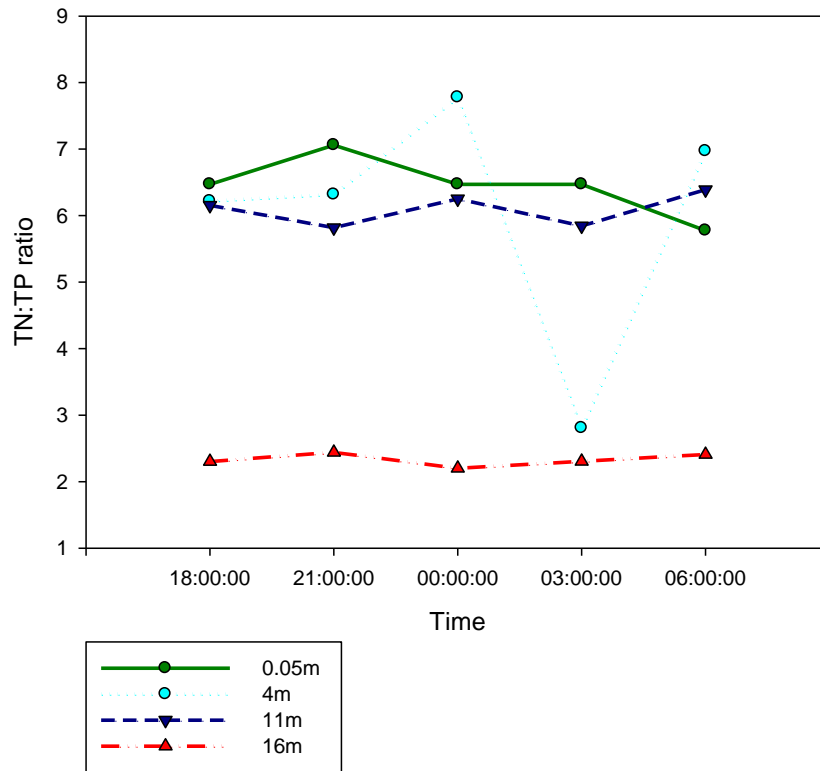


Figure 4.34 TN:TP ratios from Ida Lake nocturnal nutrient profiles, 20th January 2009

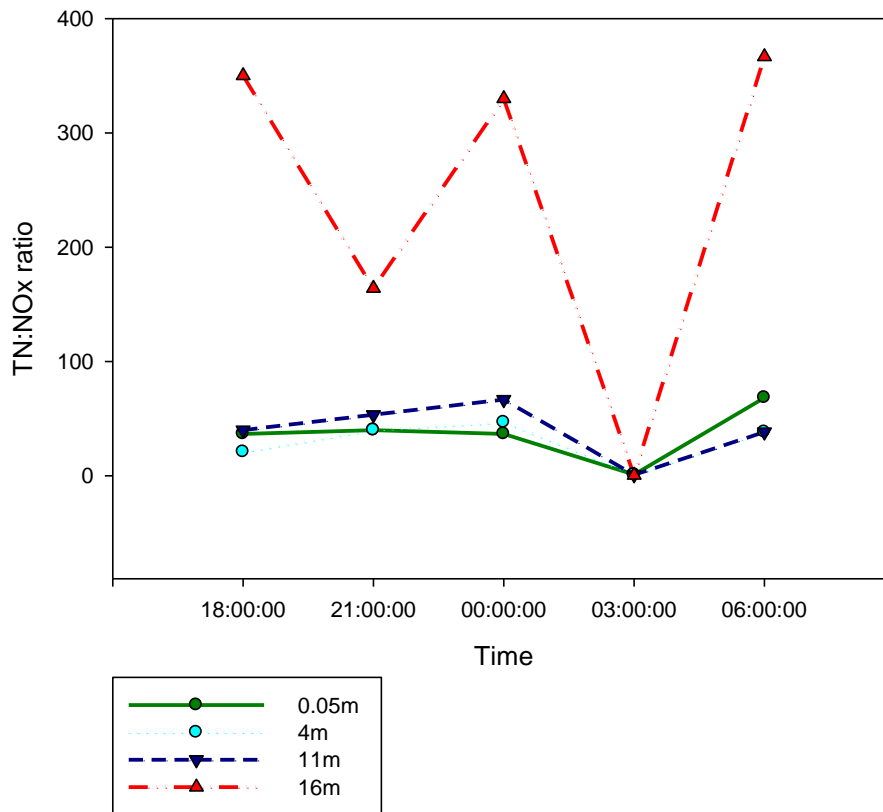


Figure 4.35 TN:NOx ratios from Ida Lake nocturnal nutrient profiles, 20th January 2009

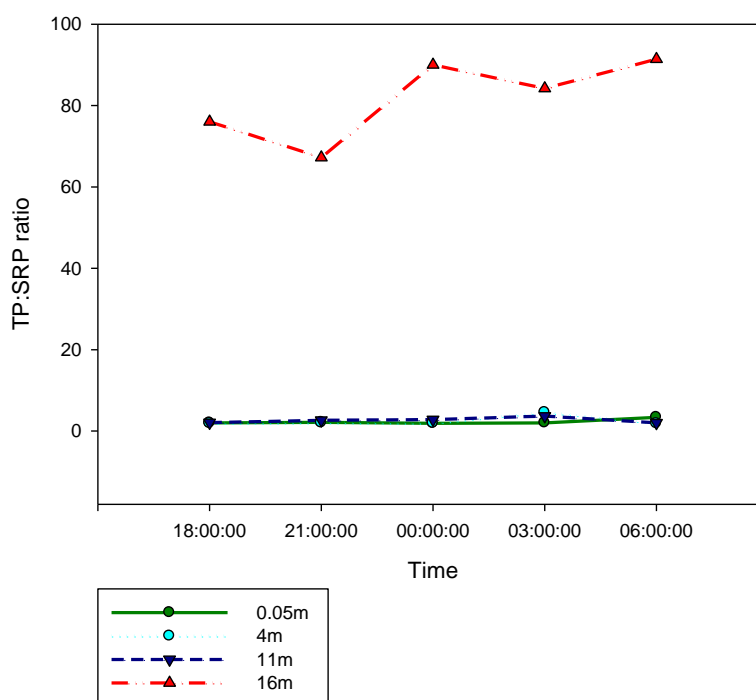


Figure 4.36 TP:SRP ratios from Ida Lake nocturnal nutrient profiles, 20th January 2009

An interesting diel pattern of TP concentrations was found at the Bootawa Dam off-take in the lower Manning River (Fig. 4.37). MCW collected nutrient samples via an automated sampler located near the Bootawa Dam off-take. These samples were collected from the mid water column at 4.30 am and 3.00 pm each day from January 2009 to February 2009. An increase, particularly in TP, during the afternoon was found when compared to samples collected in the early morning. During this low-flow period ($<5 \text{ m}^3\text{s}^{-1}$ at Killawarra), the trend appeared to be strong with variations of up to 0.011 mgL^{-1} between 4.30am and 3.00pm on the same day (Fig. 4.37). The difference between nocturnal and daytime concentrations is reduced when discharge increases (Fig. 4.37).

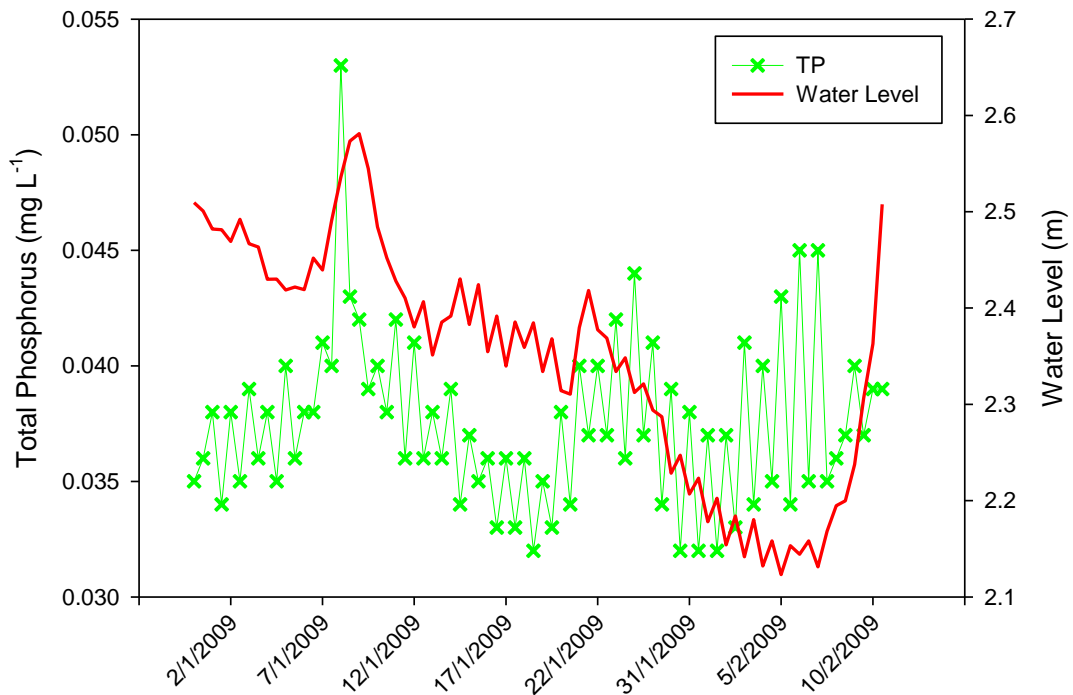


Figure 4.37 Diurnal patterns of TP concentrations at Bootawa off-take, January –February 2009
(data source: MCW)

4.3.2.2 *Shallow Pools*

The decline in water temperature with depth was closely mimicked by DO saturation (Fig. 4.38). While the shallow pool remained well-oxygenated at above 85% saturation, there is around a 20% decrease in DO at all sites within the epibenthic zone, with the greatest reductions found between 6pm and 9pm and again between 3am and 6am (Fig. 4.38). Saturation was within the regionally acceptable range of 75-110% (see Chapter 3).

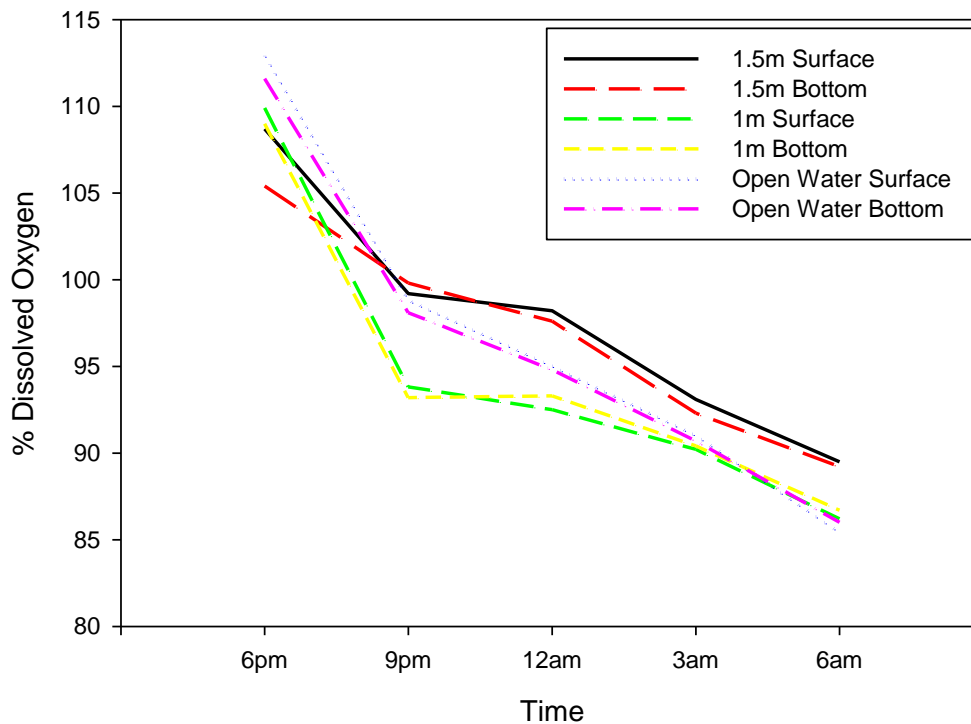


Figure 4.38 Nocturnal dissolved oxygen saturation profiles, Railbridge Pool, 4th February 2009

A nocturnal decline in pH of about 0.5 units was measured at all Railbridge sites (Fig. 4.39). All sites exhibited elevated pH levels of over 8.1 at 6pm, which is above the upper threshold of 7.5 for lowland coastal streams. As also found in water temperature and DO responses, the sites containing macrophytes displayed a plateauing or slight increase in pH between 9 pm and midnight (Fig. 4.39).

As CO₂ influences pH, the similarity between oxygen decline rates indicate that changes of oxygen influenced pH levels particularly within the macrophyte beds (Figs. 4.38 and 4.39).

There were only minor fluctuations observed in electrical conductivity within the shallow pool sites during the 12-hour sampling period and levels remained within the regionally-derived threshold.

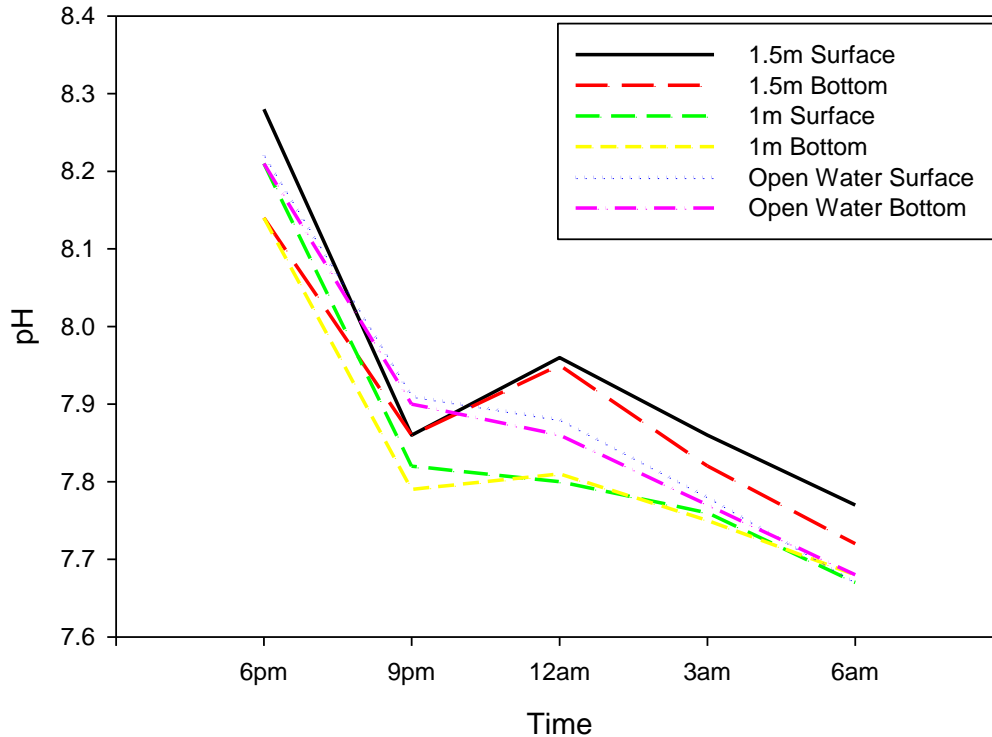


Figure 4.39 Nocturnal pH profiles, Railbridge Pool, 4th February 2009

Results showed that the regionally-derived 80th percentile threshold of 0.35 mgL⁻¹ for TN was exceeded at the surface of the 1.5 m macrophyte site at 6 pm and 9 pm, with concentrations reducing during the night (Fig. 4.40). All other N-related variables were within recommended thresholds (Figs. 4.40 and 4.41).

Total nitrogen and ammonium concentrations decreased during the night, with the greatest reduction occurring between 6pm and 12am at the Railbridge sites which contained macrophytes (Fig. 4.40 and 4.42). The open water site had much lower concentrations of TN and ammonium compared to the macrophyte sites on all sampling occasions (Figs. 4.40 and 4.41). Nitrate/nitrite concentrations were generally low or below detection throughout the sampling period, with the exception of nitrite concentrations at the epibenthic zone of macrophyte sites (Fig. 4.42).

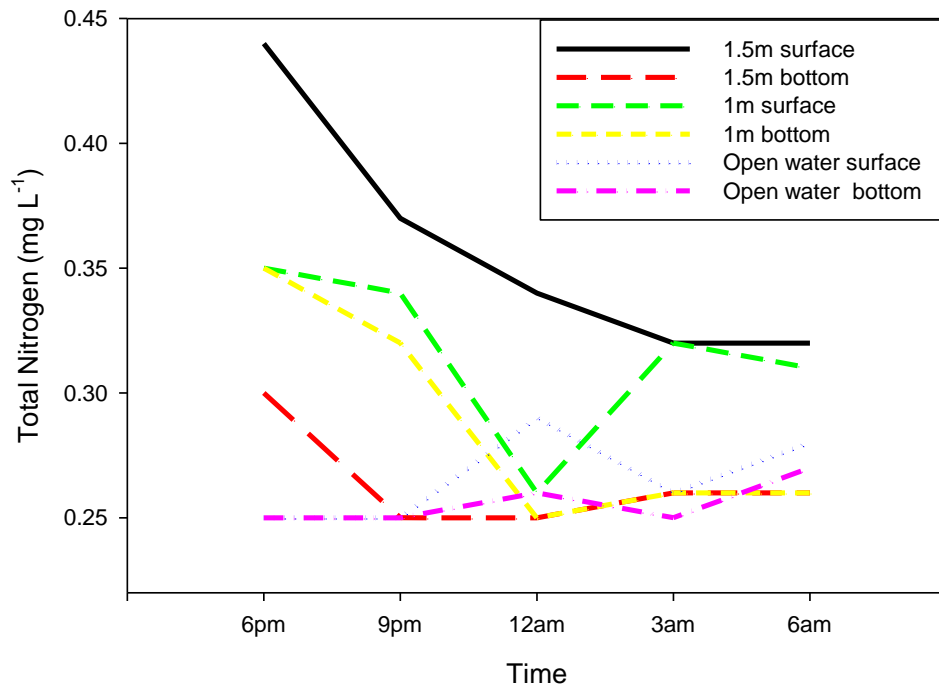


Figure 4.40 Nocturnal total nitrogen concentrations, Railbridge Pool, 4th February 2009

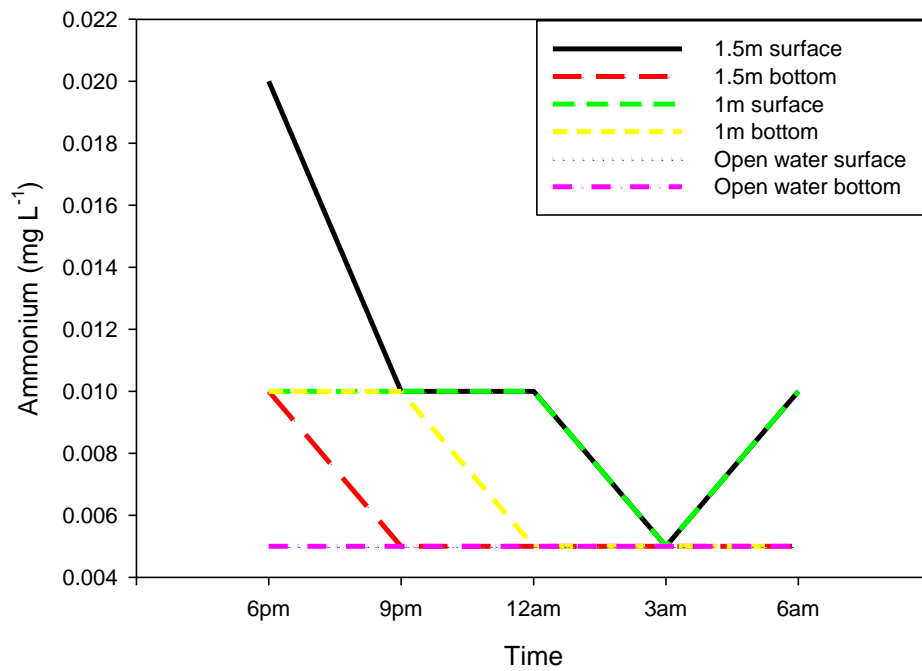


Figure 4.41 Nocturnal ammonium concentrations, Railbridge Pool, 4th February 2009

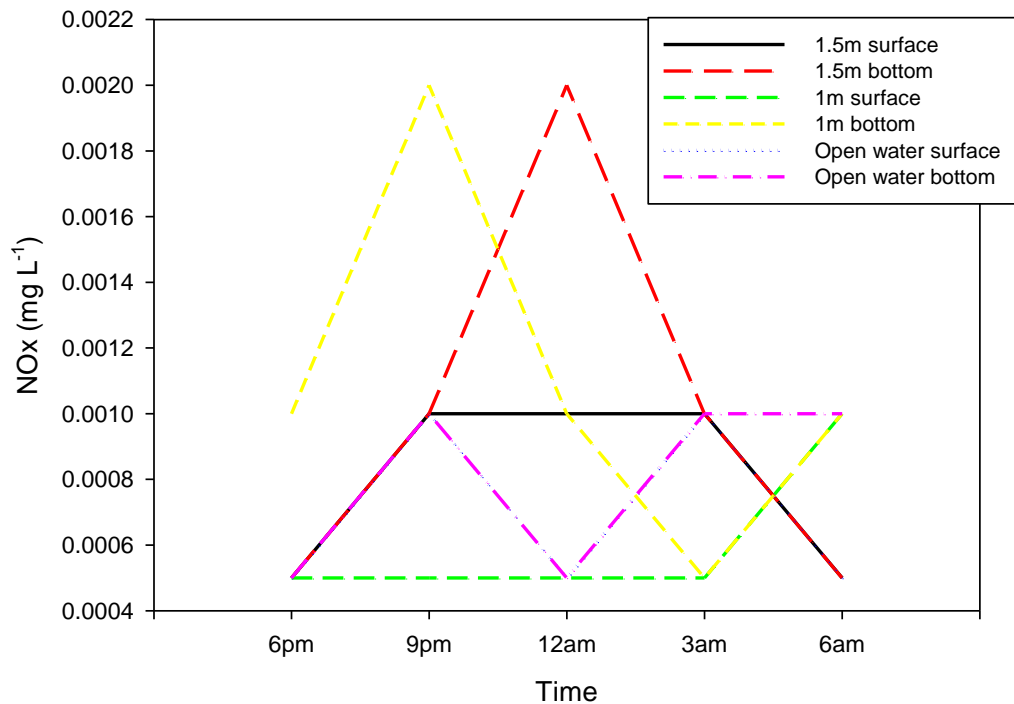


Figure 4.42 Nocturnal NO_x concentrations, Railbridge Pool, 4th February 2009

As was found with N, data indicated that differences may occur between surface and epibenthic concentrations of P within shallow macrophyte beds under low flow conditions. The recommended 80th percentile regionally-derived threshold of 0.03 mgL⁻¹ for TP was exceeded at all sites on all occasions (Fig. 4.43). The SRP threshold of 0.02 mgL⁻¹ was only exceeded once at 9pm at the 1 m bottom macrophyte site (Fig. 4.44).

Total phosphorus showed similar patterns at the two macrophyte-dominated sites, with a decline between 6 pm and midnight, followed by an increase from midnight to 6 pm (Fig. 4.43). The concentration and increase in TP was greatest at the macrophyte surface sites (Fig. 4.43). The open water site showed less variation between concentrations over time (Fig. 4.43). Soluble reactive phosphorus showed a similar trend to TP at surface sites (Fig. 4.44). However, the bottom of the 1-m macrophyte site showed a large increase in SRP at 9 pm (Fig. 4.44). The open water site had lower concentrations of SRP throughout the night (Fig. 4.44).

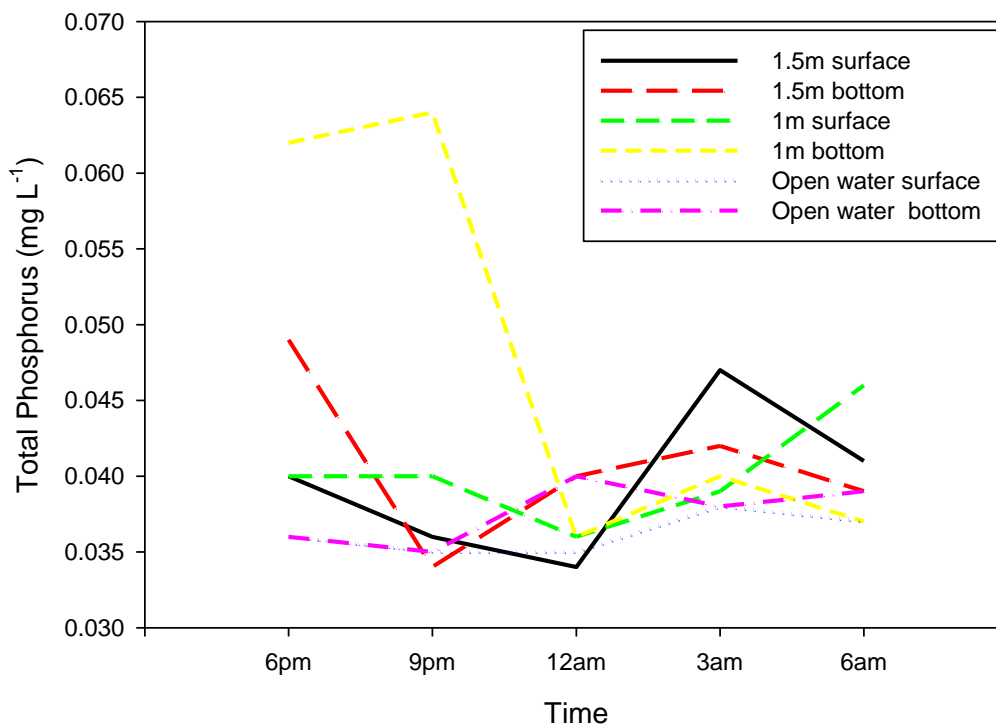


Figure 4.43 Nocturnal total phosphorus concentrations, Railbridge Pool, 4th February 2009

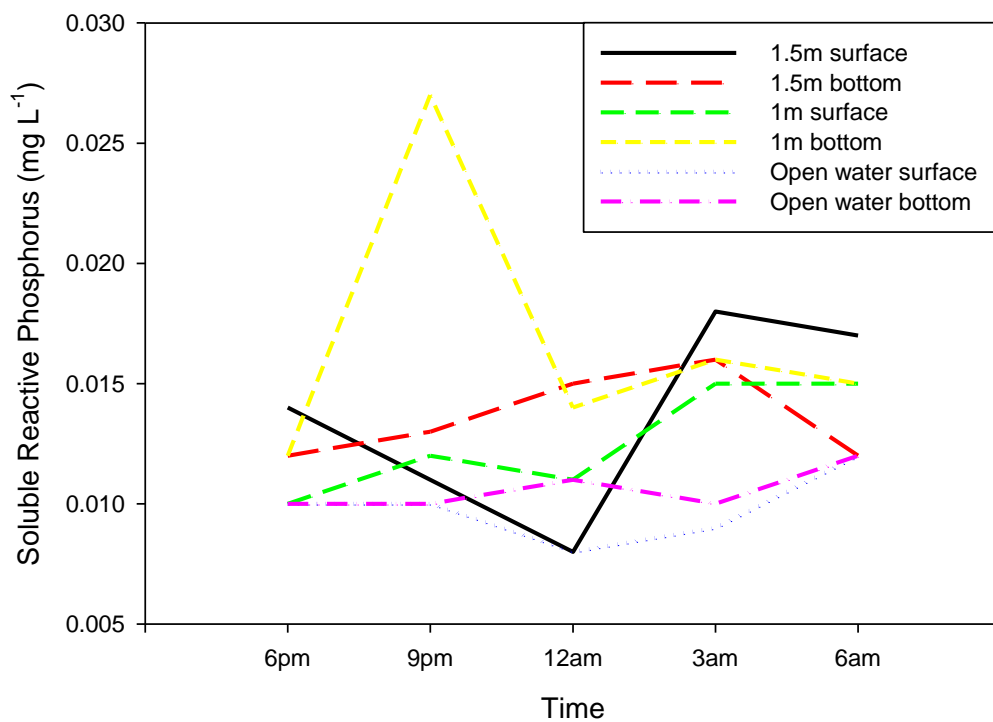


Figure 4.44 Nocturnal soluble reactive phosphorus concentrations, Railbridge Pool, 4th February 2009

Surface waters within shallow pool macrophyte beds generally supported higher ratios for TN:TP and TN:NO_x in the earlier part of the evening (18:00 to 21:00) (Figs. 4.45 and 4.46). TN:TP ratios ranged between 9:1 and 11:1 for the surface of the macrophyte beds and between 5:1 and 8:1 for other areas (Fig. 4.46). Similarly for TN:NO_x, surface macrophyte sites ranged from 70:1 and 90:1 compared to 50:1 and 60:1 for other sites during the early evening (Fig. 4.46). There was little difference between sites for TP:SRP with ratios ranging from 2.5:1 and 5:1 throughout all sites (Fig. 4.47).

When compared to Ida Lake, TN:TP ratios were similar to the Ida Lake surface ratios, less than 16:1, which may indicate N limitation (Figs. 4.34 and 4.45). TN:NO_x ratios were also within the Ida Lake surface water ranges (Figs. 4.35 and 4.45), as too was the TP:SRP ratio (Figs 4.36 and 4.47).

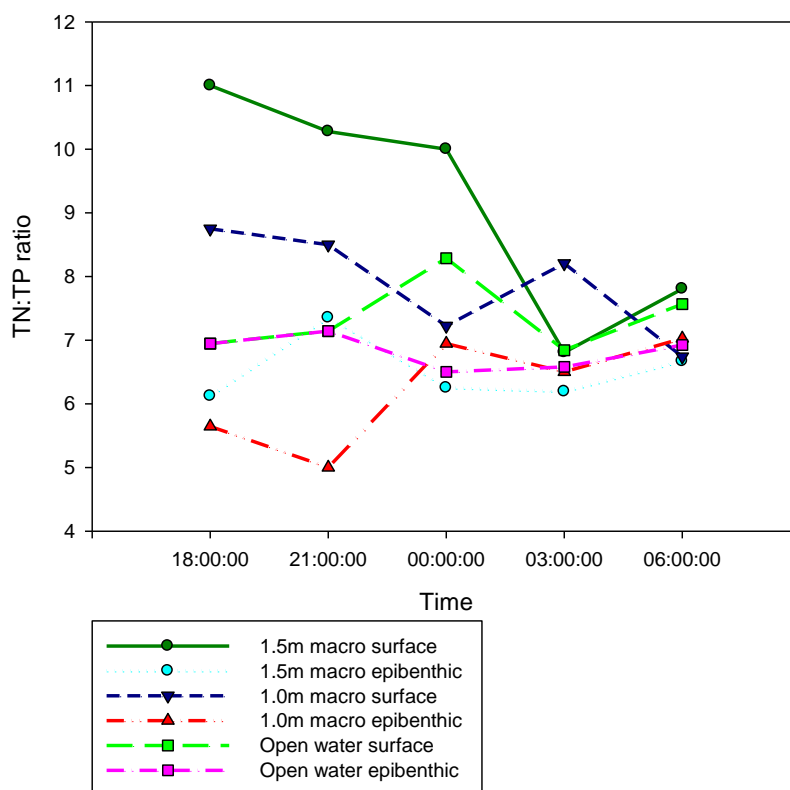


Figure 4.45 TN:TP ratios from Railbridge Pool nocturnal nutrient profiles, 4th February 2009

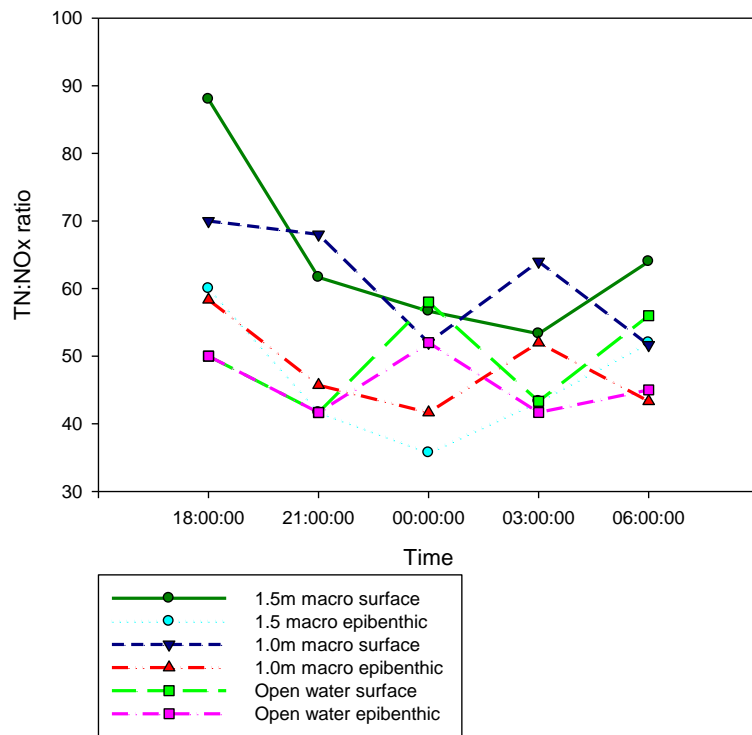


Figure 4.46 TN:NOx ratios from Railbridge Pool nocturnal nutrient profiles, 4th February 2009

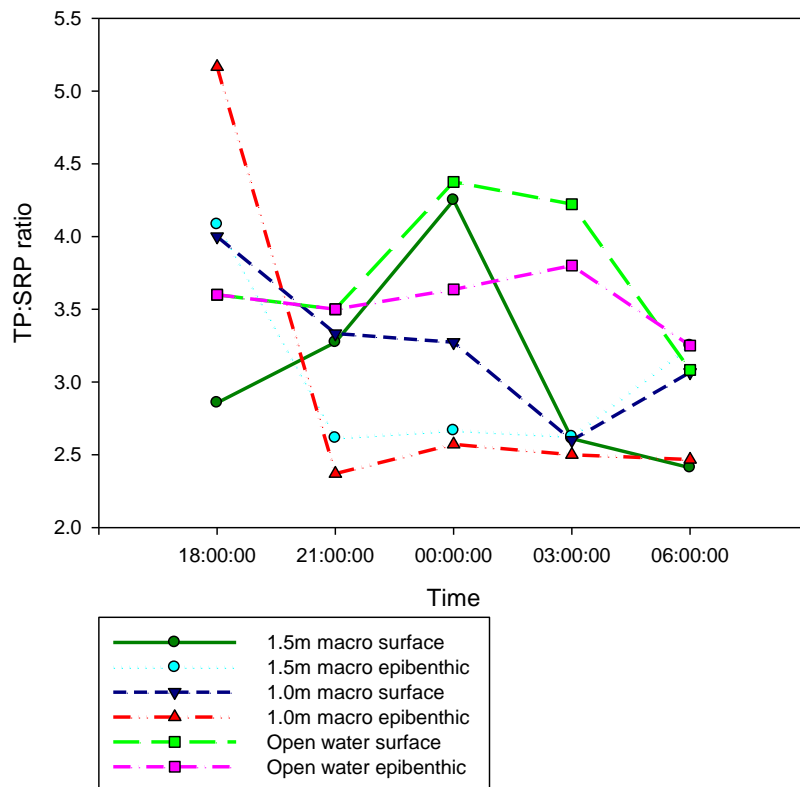


Figure 4.47 TP:SRP ratios from Railbridge Pool nocturnal nutrient profiles, 4th February 2009

4.3.3 Ecological consequences

4.3.3.1 Deep pools

The first axis of the PCA performed on nutrient, pH, dissolved oxygen and water temperature data from the deep Ida Lake (Fig. 4.48) explained 99.8% of the total variance within the similarity matrix. The second and third axes evenly accounted for the remaining variation (Table 4.3), resulting in total of 100% of variability captured by these three axes. Over 70% of total variability is considered a good target figure by Clarke and Gorley (2006).

The location of end-points in the ordination space relative to the sites, water depth and time indicate the environmental and chemical variables influencing differences between sites. The relationships between nutrients and other variables in Ida Lake indicated three strong groupings along the first axis, with outliers along the second axis. The deepest section of the pool was strongly related to increased concentrations of ammonium, total nitrogen and total phosphorus (Fig. 4.48). Samples taken from the surface and at 12 m, which is above the thermocline, showed a strong negative correlation to water temperature, dissolved oxygen, pH and filterable reactive phosphorus (Fig. 4.48). This negative correlation may reflect the influence of the thermocline interface between surface water quality and lower depths within the pool (Fig 4.48).

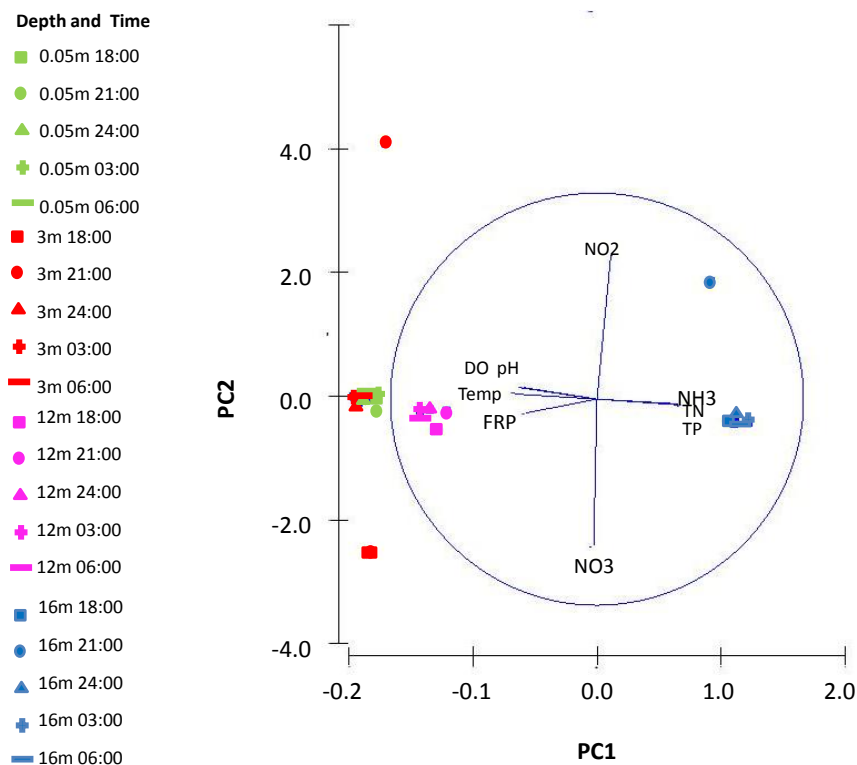


Figure 4.48 Principal Components Analysis of nutrient and environmental variables at different depths in Ida Lake, with vectors showing the contributions of variables to the axes

The vector length reflects that variable's contribution to the two axes in relation to all possible axes. The circle represents the maximum possible vector length.

Table 4.3 Coefficients of the eigenvectors in the linear combinations of variables making up the principal components, and eigenvalues and variance accounted for by each axis for Ida Lake data Coefficients for variables contributing the highest to each axis highlighted in bold.

Variable	PC1	PC2	PC3
Depth	0.147	0.895	0.413
Temp	-0.076	0.348	-0.803
DO	-0.986	0.101	0.134
pH	-0.011	-0.038	-0.023
NH ₄ ⁺	0.018	-0.188	0.289
NO ₂	0.000	0.000	0.000
NO ₃	0.000	0.000	0.000
TN	0.016	-0.160	0.258
FRP	0.000	0.001	-0.001
TP	0.007	-0.077	0.127
Eigenvalues	0.0174	2.49	1.3
% Explained Variance	99.8	0.1	0.1
Cumulative Variance	99.8	99.9	100.0

4.3.3.2 Shallow Pools

The first two axes of the PCA performed on nutrient and environmental results from the shallow Railbridge Pool (Fig. 4.49) explained 70.8% of the total variance within the data matrix (Table 4.3). The third axis accounted for 16.5% of variation each (Table 4.4) resulting in total of 87.3% of variability captured by these three axes.

When results from the PCA of nutrient and other variables from the macrophyte pool sites were analysed, there are a number of site groupings along the first and second axes. The endpoints relative to sites showed the two macrophyte sites (Sites 1 and 2), sampled at 18:00 hours had a positive association with TN, pH, water temperature and DO (Fig. 4.49). These sites had concentrations of TN of at least 0.30 mgL⁻¹ at this time (Fig. 4.40). The epibenthic sample taken from site 2 at 21:00 is an outlier along the first axis (Fig. 4.49). This site at this time had higher TP (0.064 mgL⁻¹) and SRP (0.027 mgL⁻¹) then all other sites at this time (Figs. 4.43 and 4.44). Site 3, the open water site, where macrophytes were absent, had the greatest negative relationship with these variables within the ordination space (Fig. 4.49).

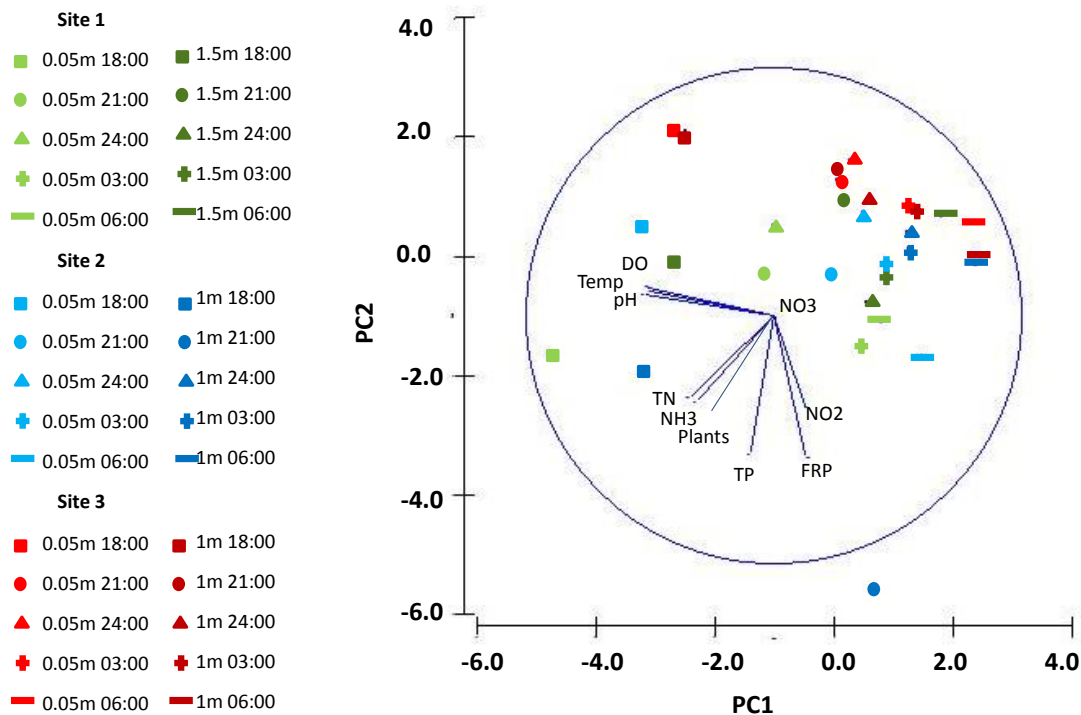


Figure 4.49 Principal Components Analysis of nutrient and environmental variables at different depths in Railbridge Pool, with vectors showing the contributions of variables to the axes.

The vector length reflects that variable's contribution to the two axes in relation to all possible axes. The circle represents the maximum possible vector length.

Table 4.4 Coefficients of the eigenvectors in the linear combinations of variables making up the principal components, and eigenvalues and variance accounted for by each axis, performed on Railbridge Pool data. Coefficients for variables contributing the highest to each axis highlighted in bold.

Variable	PC1	PC2	PC3
Temp	-0.480	0.095	-0.336
DO	-0.507	0.112	-0.246
pH	-0.517	0.084	-0.167
NH ₄ ⁺	-0.332	-0.322	0.464
NO ₂	0.126	-0.363	-0.519
NO ₃	0.000	0.000	0.000
FRP	0.127	-0.563	-0.159
TP	-0.094	-0.550	-0.206
TN	-0.306	-0.340	0.496
Eigenvalues	3.43	2.24	1.32
% Explained Variance	42.8	28.0	16.5
Cumulative Variance	42.8	70.8	87.3

4.4 Discussion

4.4.1 Pool water quality responses to low flows

Chapter 3 described the relationship between nutrient concentrations and inflows to the Manning River, with the contribution of suspended sediment and terrestrial organic material increasing during high flows. In this Chapter I describe the in-stream mechanisms which can alter nutrient concentrations under moderate to low flows. Factors such as channel morphology, flow characteristics, biological activity and temperature have major influences on nutrient speciation, rates of release and uptake (Meyer and Likens 1979, Mulholland 1996, Solimini et al. 2005, Withers and Jarvie 2008).

Under summer low flow conditions, both deep pools and shallow macrophyte-dominated pools responded with a reduction in DO concentrations and increases in nutrients. Persistent thermal stratification was recorded in deep pools under these summer low flow conditions. The influence of stratification on water quality within the hypolimnion of the deep pools, driven by anoxic conditions, included increases in EC, TN, NH_4^+ and TP, and a reduction in pH. Changes in water quality within shallow, macrophyte-dominated pools under summer low flow conditions included reductions in DO over the nocturnal sampling period, increased N within macrophyte beds and increased TP at all shallow pool sites.

The decline in oxygen concentrations under stratified conditions in Ida Lake appeared to be driven by both nocturnal convection and respiration. Other studies have shown that greater bacterial productivity occurs in the thermocline area due to interactions between products such as methane or ammonia in the anoxic hypolimnion and the oxygenated epilimnion. Harrits and Hanson (1980) found an abundance of methane-oxidising bacteria within a thermocline located in a stratified lake, as this environment provided the greatest concentrations of oxygen and methane. The presence of large numbers of bacteria in the thermocline, coupled with the lack of photosynthetic activity and nocturnal convection may have resulted in the apparent drawdown of oxygen across the thermocline of Ida Lake.

The increase in EC within the hypolimnion of Ida Lake and Bungay Pool indicates an increased interaction with groundwater or a lack of mixing within both deep pools under low flow conditions. A lack of mixing may result in bacterial decomposition within the hypolimnion converting organic matter to bicarbonate and carbonate ions dependent on the pH (Heaney et al. 1986). The resulting increase, while more than twice that found at the surface, was within ANZECC/ARMCANZ (2000) guidelines for the protection of coastal river ecosystems. The deep-water increments of conductivity also imply some increase in density, which might further enhance stratification (Heaney et al. 1986).

The lower pH found in the hypolimnion of Lake Ida, along with reduced redox potential, could influence the movement of P from sediment to overlying water, and hence on epilimnetic primary production when exchange occurs between the hypolimnion and overlying water (Heaney et al. 1986, Townsend 1999, Bakri and Chowdhury 2006). Similarly to Ida Lake, a nocturnal decline in pH was measured within the shallow pool. This decline is related to a similar decline in oxygen. Reductions in both pH and DO within the shallow pool epibenthic zone is likely to influence redox potential and facilitate the release of nutrients from sediment or transfer of nutrients to more biologically-available forms (Jaynes and Carpenter 1986).

Differences in nutrients over the sampling period were likely to be driven by factors such as changes in oxygen and pH, and increased respiration. SRP appeared to be concentrated within the Ida Lake thermocline, possibly the result of microbial activity converting abundant TP to a more available form. The higher concentrations of SRP in the thermocline may relate to microbial activity transforming the abundant TP in the hypolimnion to the more available form of SRP (Rudd 1976, Litchman et al. 2004). The greater concentration of SRP in the thermocline may also relate to possible aerobic decomposition of algae by bacteria as described by Uehlinger (1986), who found that some species of algae rapidly released most of their phosphorus as SRP when decomposing. TP doubled in concentration at the surface of Ida Lake over the 12-hour sampling period, increasing from 0.034 to 0.071 mgL⁻¹. However, at the same time TP appears to decline in the layers immediately below the surface which indicates convection processes are influencing surface TP concentrations.

Similar to TP, NH₄⁺ and TN concentrations increased in the top layer of Ida Lake by the end of the night but decreased or remained steady in the lower layers. Total nitrogen decreased during the night within all the Railbridge sites, whereas NO_x shows peaks at various times during the night at the bottom of the macrophyte sites. This indicates an increase or change in the type of microbial activity taking place in these pools and the way nutrients are used or transformed. ORP values above 200 mV indicate an oxidising environment which can result in the conversion of ammonium to nitrates, while lower values, particularly less than 100 mV, indicate a reducing environment, which may lead to the production of ammonium and methane (Burger et al. 2007, Ozkundakci et al. 2011) (Fig. 4.51).

The mechanism for a change in concentration or form in N may also be related to N-fixation by algae or activity by ammonia-oxidising bacteria which may increase during darkness. Rudd et al. (1976) also found that methane oxidizers in the thermocline accessed nitrogen from both nitrogen fixation and diffusion of ammonia from the steep concentration gradient formed by the hypolimnion. The reduction may also relate to diurnal rates of change in nitrification and denitrification rates in rivers. A study by Laursen and Seitzinger (2004) found that both nitrification (microbially mediated oxidation of ammonium to nitrite) and denitrification rates (the microbially mediated reduction of nitrate to nitrogen gas) were consistently higher during the day than during the night. They related the diurnal changes to cyclic patterns in water column pH and temperature.

Nitrification can only occur in the presence of oxygen. Conversely, denitrification is favoured under anoxic conditions. Studies have found an association between diurnal fluctuations in oxygen and denitrification rates in sediments colonized by benthic microalgae (e.g. Christensen et al. 1990, Nielsen et al. 1990, Risgaard-Petersen et al. 1994). Other studies have linked diel changes in oxygen consumption to patterns of nitrous oxide (N₂O) and nitrogen gas (N₂) production (Laursen and Seitzinger 2004, Harrison et al. 2005). Another process through which nitrogen changes forms is microbial reduction of nitrate to ammonium through dissimilatory and assimilatory pathways (Tiedje 1988). The dissimilarity pathway is sensitive to oxygen preferring anaerobic conditions, while the assimilatory pathway is not (Tiedje 1988). The higher concentration of ammonium at the macrophyte sites in early evening suggests that assimilation of nitrate under oxygenated conditions may be occurring with producing ammonium. The concentration of ammonium had reduced at these sites by 9 pm, with nitrite concentrations

increasing at the bottom of the macrophytes sites at various stages during the night, possibly indicating nitrification.

The interaction between temperature and metabolic changes in Ida Lake, as evidenced by DO concentrations, provide different nutrient responses in the thermocline as an interface between the epilimnion and hypolimnion. This interaction is explained by Imberger and Patterson (1989) who proposed a temperature structure model, with a diurnal surface layer, a parent thermocline, a metalimnion and a hypolimnion. The parent thermocline is the change in temperature caused by the most severe mixing or deepening event in the immediate past, while the metalimnion is the region between the parent thermocline and a depth where the temperature profile reaches a certain percentage of the coldest temperature in the hypolimnion. In the case of Ida Lake, diurnal processes alter the interactions between the surface layer, parent thermocline and hypolimnion resulting in dramatic changes in nutrient concentrations and ratios between those nutrients.

The results from Bootawa Dam offtake at 12-hour intervals demonstrate an increase in TP during the afternoon when compared to the early morning sampling, particularly at low flows. This study supports the Bootawa Dam data, suggesting there may be defined periods of higher nutrient availability. It may be preferable to pump from the River to the Dam during early morning rather than the afternoon under low flow conditions. It would assist the decision-making process to recognise nutrient concentrations vary considerably over a 24 hour period and diurnal changes alter both the availability and amount of nutrients with the system. This study also suggests that, by relying solely on grab samples for water quality information, there is an overall underestimation of SRP and possibly and overestimation of TP, TN and NH_4^+ within the lower Manning.

4.4.2 Interactions between summer low flows, primary production and nutrient cycling

The Manning River nocturnal study indicated that changes to biological activity that occur during the night can result in an alteration in basic water quality variables and nutrient behaviour, in both deep pools and shallow macrophyte-dominated pools. Both study sites exhibited declines in oxygen during the night, however the degree and timing of these changes varied. These temporal changes may be caused by convection processes which are influenced by turbulence (e.g. wind or flow) or can be produced by the loss of heat from the upper layers of water due to water-air interactions (Talling 2004). As there was little wind during the nocturnal study period and flow remained steady, changes in oxygen at the water's surface are more likely to have been related to the loss of heat from surface layers affecting convection processes. This is supported by evidence of nocturnal convection leading to changes in gas transfer from other studies (Ford et al. 2002). Increases in respiration rates during the night are also likely to be a major influence on oxygen concentrations (Venterink et al. 2003).

The reduction in DO saturation at all depths within the Railbridge Pool macrophyte beds could be associated with macrophyte and algal respiration, and changes within sediment at the plant rhizosphere level. A study looking at the influence of macrophytes on nitrification and denitrification found that oxygen microprofiles around the root area indicated that oxygen was released to the surrounding sediment during both light and dark phases (Risgaard-Petersen and Jensen 1997). However, without directly measuring these rhizosphere processes, it is difficult to

determine the mechanism which may be affecting the rate of oxygen decline within the Railbridge Pool.

Reduced oxygen concentrations at both pool sites over the 12-hour sampling period were mimicked by a minor decline in pH. Biological activities, such as photosynthesis and respiration, can influence pH due to changes in CO₂ (Paerl and Ustach 1982). During daylight hours, while plants photosynthesize, pH levels usually increase, at night during respiration pH levels generally fall. As only the euphotic zone in Ida Lake was likely to have high levels of algal activity, changes in pH were restricted to this zone at night as algae respired. Within the shallow pool, the similarity between oxygen decline rates, indicate that the changes of oxygen influenced pH levels particularly within the macrophyte beds. Macrophyte-mediated changes to variables such as pH influences both abiotic and biotic nutrient processes (Clarke 2002). Changes in nutrient concentration, speciation and ratios through these interactions may impact on the flow and balance of energy in ecological interactions within features such as deep pools and in downstream environments (Frost et al. 2002, Giling et al. 2012).

Low N:P ratios (i.e. <16:1) found in the Manning infers that N may be a limiting nutrient within the system (Bothwell 1985, Neill et al. 2001). Nutrient uptake within streams is affected by nutrient ratios, with increased nutrient supply not necessarily resulting in increased uptake (Stelzer and Lamberti 2001, Schade et al. 2011) as some streams may require higher or lower ratios, depending on the limiting nutrient, to facilitate uptake (Schade et al. 2011). Changes in nutrient ratios as a result of physical and biological interactions can therefore alter uptake rates, thereby impacting on periphyton communities and nutrient availability within streams (Sardans et al. 2012).

Nutrient ratios can further influence periphyton community structure as different plant species require different relative concentrations of essential nutrients for growth and reproduction. Resource-ratio theory assumes that plants compete for nutrients, and this theory has been applied to benthic algal species (Francoeur et al. 1999, Stelzer and Lamberti 2001, Hall et al. 2005, Schade et al. 2011) and may influence downstream riffle communities.

Alterations to elemental ratios can affect stoichiometric interactions between primary producers and their consumers, favouring some taxa, resulting in changes to community structure and impacting on nutrient fluxes (Stelzer and Lamberti 2001, Singer and Battin 2007). Stelzer and Lamberti (2001) found that the community structure of benthic algae, particularly diatoms, was substantially altered by changes in N:P ratios. This suggests that periphyton community structure is sensitive to the relative proportions of nutrients within streams, and changes to this balance that occur through physical, chemical and biological interactions, can impact on this structure and dependent consumers. The consequences of elemental imbalances between instream producers and their consumers include changes in consumer growth rates, influences on food-web structure and impacts to nutrient recycling (Stelzer and Lamberti 2001, Frost et al. 2002, Giling et al. 2012).

Just as nutrients influence periphyton community structure, plants in turn can influence nutrient behaviour. Prolific macrophyte and algal growth, particularly under warm, low flow conditions, can result in changes in nutrient availability through plant influences on sediment release and organic decomposition (Chambers and Prepas 1994, Wigand et al. 2001). This may explain the general increase in TP within the shallow pool macrophyte sites over the 12-hour sampling period. TP concentrations from the surface waters of macrophyte beds showed a significant

decline at the beginning of the night, with increases towards the end of the night. These increases in TP at the water surface indicate a link with phytoplankton behaviour.

The impact of macrophytes on nutrient dynamics in the lower Manning also appears to be partitioned, albeit a much different and less defined way than that found in a deep stratified pool such as Ida Lake. The exchange of nutrients across the sediment-water interface occurs in a much more dynamic way as environments surrounding macrophyte rhizomes have been shown to alter sediment chemistry through oxidation (Risgaard-Petersen and Jensen 1997, Howard-Williams 1985). The influence of submerged macrophyte rhizomes on bacterial nitrification and denitrification has been shown to increase nitrogen removal from the surrounding sediment (Risgaard-Petersen and Jensen 1997), while phosphorus release has been shown to be significantly higher in macrophyte-dominated areas (Stephen et al. 1997). Within the water column, photosynthetic and decomposition processes that accompany the presence of a macrophytes and the associated epiphytic and microbial communities can significantly influence instream nutrient dynamics (Clarke 2002). While nutrient uptake by growing macrophytes via the sediment and water column can be substantial, decaying macrophytes have been found to be significant instream contributors of nutrients (Clarke and Wharton 2001, Withers and Jarvie 2008).

Carlton and Wetzel (1988) found that algae on sediments can significantly influence P dynamics at the sediment-water interface. During daylight, surface sediments became oxygenated inhibiting phosphorus release from deeper sediments. During the night, microzones within the surface sediment can become anoxic and P may be released to overlying water at an accelerated rate which can produce a marked diel change (Carlton and Wetzel 1988).

The changes in P within the epibenthic zone of the macrophyte beds may relate to the effect of oxygen release from the roots of submersed vascular plants influencing ORP and therefore sediment solute dynamics (Jaynes and Carpenter 1986). Significant differences in pH and ORP potential in sediments which contain macrophytes have been found to greatly influence the availability of P and Fe fractions among sediments (Jaynes and Carpenter 1986), with streams shown to undergo complete reduction and oxidation sequences in just a few hours (Harrison et al. 2005). Added to this are diel influences including changes in respiration rates which have been found to alter factors such as pH, DO and ORP.

Although ORP was not measured during the nocturnal study, routine sampling indicated a highly oxidising environment during daylight hours over the warmer months within the macrophyte dominated pools, with values exceeding 350 mV at all sites and depths. ORP values above 200mV indicate an oxidising environment, while lower values indicate a reducing environment, which may lead to the remobilization phosphorus from the sediments back into the water column (Townsend 1999, Bakri and Chowdhury 2006) (Fig. 4.50). As pH levels are usually greater than 7, it is likely that when ORP drops below 200mV a basic reducing environment develops (Fig. 4.50). Below a redox potential of 200 mV, the mobility of P increases, as Fe (III) is reduced to Fe (II) and both iron and P are brought into solution (Mortimer 1941, 1942).

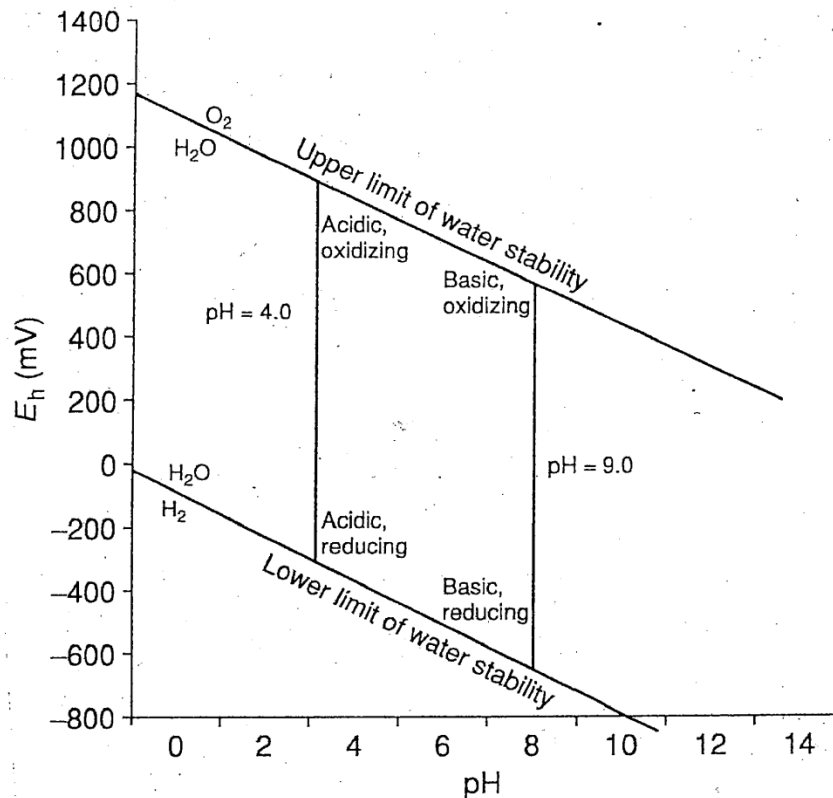


Figure 4.50 Redox potential and pH range found in soil surface environments (Reproduced from DeLaune and Reddy 2005)

The increase in TP concentrations at the surface of Ida Lake between 3 am and 6 am could be associated with the vertical movement of phytoplankton, particularly at night. Previous studies looking at the vertical distribution of inorganic phosphorus in lakes have shown that increases in P across the thermocline becomes available to phytoplankton (Salonen et al. 1984). To take advantage of this P source, motile alga regularly migrated vertically, moving below the thermocline at night and returning to the surface waters in early morning, often across a 10°C temperature gradient (Salonen et al. 1984). When experimental tubes were used to look at the upward movement of P between the hypolimnion to epilimnion it was attributed to transport by phytoplankton cells undertaking active vertical migrations (Salonen et al. 1984).

In-stream contributions to nutrients through sediment, plant and microbial interactions within Manning River pools are an important consideration. It appears that these interactions can considerably influence nutrient types and concentrations under summer low flow conditions. Therefore, it is important to recognise that nutrient concentrations are not just influenced by runoff and other inflows, but also in-stream processes which may have a major impact on concentrations and may alter nutrient availability within a short time-frame.

4.4.3 Comparisons with water quality guidelines

Chapter 3 of this thesis indicated that nutrients exceeded regionally-derived 80th percentile threshold values at non-reference sites particularly under high flow conditions. When looking at the response to low flow conditions, particularly within pools, resulting stratification has been

found to influence nutrient concentrations and forms, often greatly exceeding regionally-derived thresholds.

A preliminary investigation, undertaken prior to this thesis, looked at the behaviour of nutrients within pools under low-flow conditions (Thurtell and Bishop 2006). This investigation found that TN and NH_4^+ concentrations in the hypolimnion of Ida Lake were much higher than regionally-derived thresholds (0.35 mgL^{-1} for TN and 0.02 mgL^{-1} for NH_4^+ Chapter 3) (Fig. 4.51). TN was found to be between 4 to 5 times greater than 0.35 mgL^{-1} , while NH_4^+ was more than 80 times greater than the regionally-derived threshold for lowland coastal streams. Nitrogen concentrations in the epilimnion met nitrogen thresholds (Fig. 4.51), with NO_x meeting recommended thresholds at all depths (Fig. 4.42).

Nitrogen concentrations from the shallow macrophyte-dominated pool were generally within regionally-derived 80th percentile thresholds, with one nocturnal TN concentration exceeding the threshold at the surface of the deep macrophyte site. This exceedance was short-term, with concentrations reduced 3 hours later. Riis et al. (2012) found that macrophyte-dominated habitats had four times higher primary uptake rates for N compared to non-macrophyte habitats. This uptake rate reflects the important role macrophytes play in the retention of N and the reduction in N transport downstream particularly during the growing season compared to non-macrophytes areas (Riis et al. 2012).

Similarly to N, profile measurements of P and SRP undertaken in Ida Lake under low flow conditions during a preliminary investigation in 2007 indicated that TP levels in the hypolimnion were over 60 times higher than regionally-derived thresholds for lowland coastal rivers (0.03 mgL^{-1} for TP and 0.02 mgL^{-1} for SRP Chapter 3) (Fig. 4.52). These increased nutrients within stratified pools become available to downstream environments following stratification breakdown, the impact of these nutrients depends on nutrient uptake efficiencies and hydrologic connections between nutrient sources and sinks (Powers et al. 2012, Baron et al. 2013). The influence of this pulse of nutrients on aquatic biota is to increase downstream nutrient availability and potential primary productivity (Davis and Koop 2006).

TP concentrations within the shallow macrophyte-dominated pool exceeded thresholds at all sites on all occasions. At times, TP exceedances were more than twice that of the regionally-derived threshold. The SRP threshold was exceeded once at the epibenthic zone of the shallow macrophyte site. These exceedances are consistent with the generally high P concentrations measured throughout the lower Manning. The spatial and temporal variations found throughout the nocturnal study also highlighted the importance of biological interactions between primary producers and nutrients. High P loading has also been found to increase N loss from the water column, possibly as a result of plant uptake and greater sedimentation within macrophyte beds (Olsen et al. 2015). The N concentrations remaining within regionally-derived thresholds in comparison to P concentrations within the macrophyte-dominated pools of the Manning River may be a reflection of these processes.

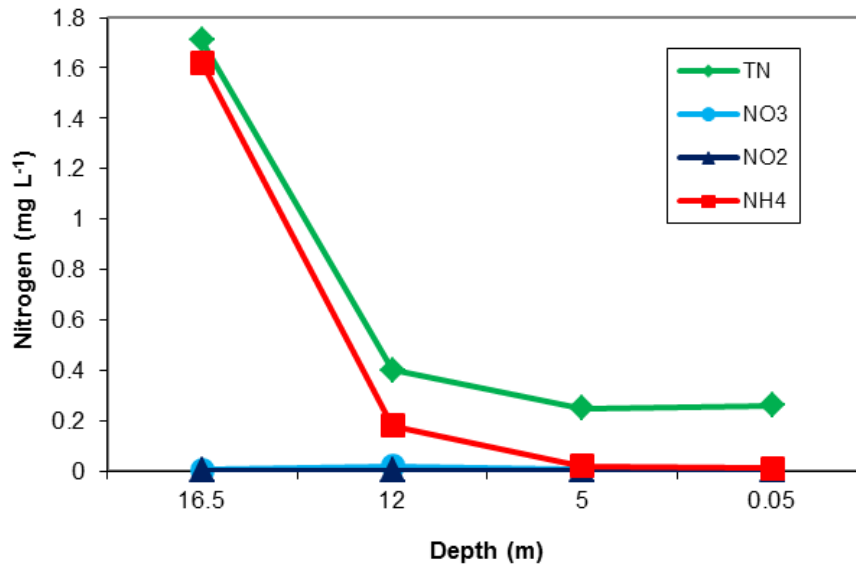


Figure 4.51 Nitrogen concentrations in Lake Ida under low flow conditions, January 2007

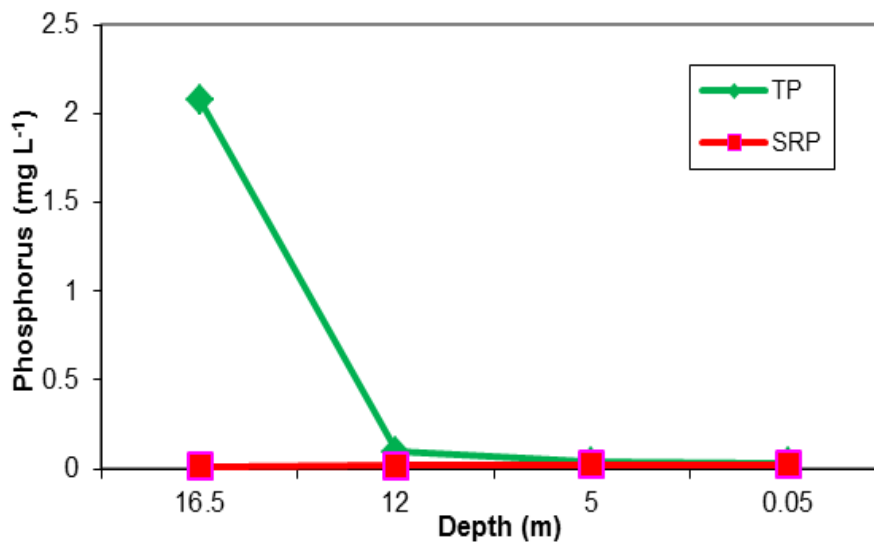


Figure 4.52 Phosphorus concentrations in Lake Ida under low flow conditions, January 2007

4.4.4 Conclusion

Summer low flow conditions resulted in thermal stratification and lower DO saturation at depth in deep Manning River pools. The reduction in flow and low DO within deep pools increased nutrient concentrations, particularly in the hypolimnion under anoxic conditions. Coupled with physical changes under summer low flow conditions such as increased ground water influences and water temperature, other interactions such as nutrient release from pool sediments and microbial processes within the hypolimnion, altered the concentrations and speciation of N and P.

Interactions between primary productivity, summer flows and water temperature also influenced DO, pH and nutrient behaviour in shallow-macrophyte-dominated pools. Low DO through increased oxygen consumption and reduced replenishment supported a reducing environment increasing nutrient concentrations, particularly TP, within the water column through sediment-nutrient release.

Regionally-derived nutrient thresholds were exceeded by all measured nutrients, with the exception of SRP, within the hypolimnion of deep pools. While there was some exceedance of N thresholds within the shallow macrophyte-dominated pools, it was TP thresholds which were exceeded at all sites and throughout the sampling period. These results emphasise the importance of short-term and seasonal changes in nutrient concentrations. The transport of nutrient-rich water from pool area to downstream environments may result in adverse effects particularly on macrophyte and algal communities and associated benthic communities. The degree of possible impacts on downstream aquatic communities is an area which needs further investigation within the lower Manning River.

The influence of summer low flows on water quality within the lower Manning should be considered when reviewing current extraction rules. There are significant implications, not only on possible impacts instream communities but also Bootawa Dam nutrient status, as a result of the spatial and temporal increases in nutrient concentrations and availability under low flow conditions. To better inform water sharing plan rules, improved understanding of the influence of water quality changes on the lower Manning River under varying flow and seasonal conditions is needed.