



Gippsland Lakes Environmental Study

Technical Reports

Further Analysis of Sediment Core Samples Collected in the Gippsland Lakes – 2000, prepared by Ian T. Webster and Brett Wallace, CSIRO Land & Water

Isotherm Analysis of Sediment Samples and Water Column Samples Collected in the Gippsland Lakes – 2000 & 2001, Prepared by Ian T. Webster, CSIRO Land & Water and Michael Grace, Water Studies Centre, Monash University

Examination of Water Balance Scenarios for Locks Across Lakes Entrance and McLennans Strait, prepared by Ian T. Webster, CSIRO Land & Water

First-cut Estimation for Seepage across Bunga Arm Barrier, prepared by Kim Seong Tan, Centre for Environmental Applied Hydrology, Department of Civil & Environmental Engineering, The University of Melbourne

Pre-European Load Estimates into the Gippsland Lakes, prepared by Rodger Grayson, KS Tan, and Stephen Wealands, Centre for Environmental Applied Hydrology, Department of Civil and Environmental Engineering, University of Melbourne

Note: This collection of five technical reports from the Gippsland Lakes Environmental Study has been collated for convenience. Individual reports should be referenced separately.

Table of Contents

FURTHER ANALYSIS OF SEDIMENT CORE SAMPLES COLLECTED IN THE GIPPSLAND LAKES – 2000	4
Data Collection	4
Evaluation of pore-water mixing rates	6
Sedimentation Rate	11
Long-Term Release of Phosphorus and Nitrogen from Sediments	12
Summary	21
References	22
ISOTHERM ANALYSIS OF SEDIMENT SAMPLES AND WATER COLUMN SAMPLES COLLECTED IN THE GIPPSLAND LAKES – 2000 - 2001	23
Introduction	23
Benthic Sample Analyses	23
Isotherm Fitting Procedure	25
Suspended Sediment Analyses	30
Isotherm Analysis on Suspended Sediment Samples	30
Conclusions	33
References	34
Appendix - Short Progress Report on P Sorption & Particle Sizing	35
EXAMINATION OF WATER BALANCE SCENARIOS FOR LOCKS ACROSS LAKES ENTRANCE AND MCLENNANS STRAIT	37
Water Balance Equation	37
McLennans Strait Lock	37
Lakes Entrance Lock	38
Discussion	40
References	40

FIRST-CUT ESTIMATION FOR SEEPAGE ACROSS BUNGA ARM BARRIER	41
Introduction	41
Seepage through the Bunga Arm Barrier	43
Discharge through Lakes Entrance Waterway	44
Significance of Seepage	44
Conclusion	45
PRE-EUROPEAN LOAD ESTIMATES INTO THE GIPPSLAND LAKES	46
Overview	46
Methods	46
Results	47
Discussion	48

Further Analysis of Sediment Core Samples Collected in the Gippsland Lakes – 2000

Ian T. Webster and Brett Wallace
CSIRO Land & Water

The Gippsland Lakes Environmental Study centres on the development of a biogeochemical model that describes the fates of nitrogen (N) and phosphorus (P) within the system and the likely impacts of the transport and transformation of these nutrients on the primary production within the Lakes. A key component of the biogeochemical model is the sediment module that describes the diagenesis of organic matter that has settled from the water column. Some portion of the nutrients within this organic matter is recycled to the water column to contribute again to primary production there. We expect that a proper description of the cycling of nutrients between water column and sediments is crucial if the biogeochemical model is to function properly.

The purpose of this report is to extend the analysis of data collected by Longmore (2000a) to support the components of the Study concerned with sediments. Specifically, Longmore measured profiles of a number of chemical species within the sediments at sites throughout the Lakes. In this report, we use these data to address objectives that are crucial to the development of the biogeochemical model. These objectives are to:

1. Estimate pore-water mixing rates that can be applied to the full Lakes' biogeochemical model.
2. Estimate sedimentation rates throughout Lakes.
3. Assess the likely extent of continued release of P and N from the stored sediment pool after input loads of these nutrients to the Lakes are reduced.

Data Collection

Sediment cores were collected by Marine and Freshwater Resources Institute (MAFRI) from 21 sites throughout the Gippsland Lakes in May 2000 (Fig. 1). Table 1 provides a brief description of the sediments at each site and water depths. Single cores 10 cm in diameter and up to 30 cm deep were collected by SCUBA from 20 sites, while triplicate cores were collected from the 21st site. Cores were sectioned under a nitrogen atmosphere, pore waters separated by centrifuging, and the filtrates and solid phase analysed for a range of nutrients. Details of the sample collection and analyses are described by Longmore (2000a). The samples were analysed for % water content, sulfate (SO₄), chlorinity (Cl), ammonium (NH₄), oxidised nitrogen (nitrate + nitrite, NO_x), dissolved organic nitrogen (DON), phosphate (PO₄), silicate (SiO₄), solid-phase nitrogen (PN), solid-phase phosphorus (PP), and solid-phase organic carbon (SOC).

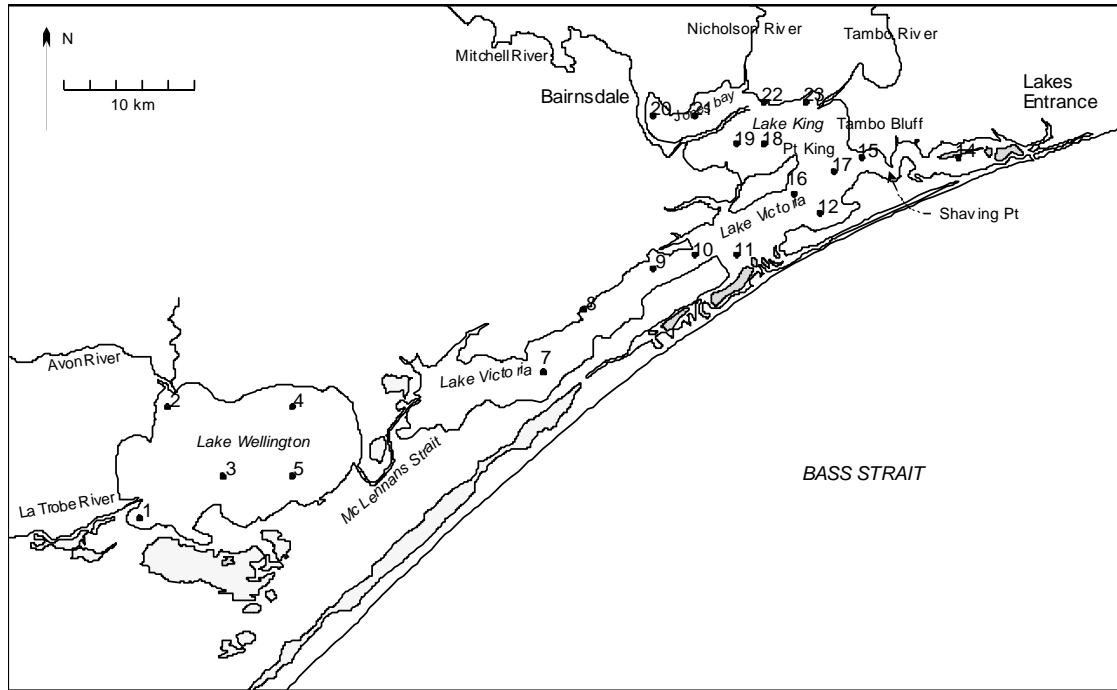


Figure 1. Locations of the sampling sites (from Longmore 2000a).

Table 1. Site locations, brief description and water depth (from Longmore 2000a)

Date	Site no.	Description	Water depth (m)
2/05/2000	1	Lake Wellington. Fine mud, lots of tubes. Top ~ 5 cm brown, grey below. Fibrous material in deeper (~20-30 cm) sediment.	1.6
2/05/2000	2	Lake Wellington. Top ~ 5 cm brown, grey below. Very stiff clay/mud below 20 cm depth.	2.0
2/05/2000	3	Lake Wellington, similar to site 2.	3.1
2/05/2000	4	Lake Wellington, similar to site 2.	2.9
2/05/2000	5	Lake Wellington, similar to site 2, but with coarser sediment below 20 cm depth	3.1
2/05/2000	6	Lake Victoria. Muddy sand, grey-black. Not analysed.	1.9
2/05/2000	7	Central Lake Victoria. Black mud throughout. Diver reported anoxic bottom water.	5.1
3/05/2000	8	Central Lake Victoria. Grey-black mud throughout.	5.4
3/05/2000	9	Eastern Lake Victoria. Grey-black mud throughout.	7.3
3/05/2000	10	Eastern Lake Victoria. 1 cm layer of black floc over black mud.	9.0
3/05/2000	11	Southern Lake King. Grey-brown fine mud, numerous 1-2 cm long isopods on surface	4.2
3/05/2000	12	Southern Lake King. Coarse sand, with cratered surface. Numerous small bivalves, and organic skin (possibly micro-phytobenthos).	2.2
4/05/2000	13	Coarse sand, with cratered surface. Not analysed.	1.2
3/05/2000	14	Off Reeves Channel. Mud/sand/mud layers: Seagrass in top 1 cm layer of fine mud on top of coarse sand at 1-2 cm, over mud.	2.1
3/05/2000	15	Southern Lake King. 1 cm brown floc over 5 cm brown/grey sandy mud grading to grey mud below.	6.5
4/05/2000	16	Southern Lake King. Grey-black mud. Surface covered with brown algal mat/slime skin.	7.0
4/05/2000	17	North Lake King. As for site 16.	6.4
4/05/2000	18	North Lake King. Grey-black sandy mud, with worms and snails at surface.	3.6
4/05/2000	19	Northern Lake King (Eagle Point Bay). Fine grey-black mud with shell fragments.	5.0
4/05/2000	20	Jones Bay. Muddy sand, grey-black, stiff below ~ 5 cm.	1.8
4/05/2000	21	Jones Bay. Muddy sand, brownish top 3 cm, grey-black below.	2.0
4/05/2000	22A	Northern Lake King. Muddy sand, brownish top 3 cm, grey-black below. Triplicate cores (A,B,C) collected within about 10 m of each other.	5.5
4/05/2000	22B	Muddy sand, brownish top 3 cm, grey-black below. Chemical properties different to 22A and 22C.	5.5
4/05/2000	22C	Muddy sand, brownish top 3 cm, grey-black below	5.5
3/05/2000	23	North Lake King. Grey-black mud throughout, with shell fragments in top 5 cm.	6.1

Evaluation of pore-water mixing rates

We use an inverse technique to estimate the rate of pore-water transport of solutes within the sediments of the Gippsland Lakes. Profiles of chlorinity were measured by Longmore (2000a) at each of the 21 sampling sites. In general, the chlorinity profiles were non-uniform along their lengths. We attribute the measured chlorinity profiles to be due to variations in the salinity (and hence chlorinity) in the overlying water which propagate downwards by pore-water transport processes. In other words, the measured chlorinity profiles are a consequence of the time-varying salinity in the overlying water and of the nature and strength of pore-water transport. We have measurements of the overlying salinity, but pore-water transport is unknown. However, we develop simulated chlorinity profiles using measured overlying salinities in a model of solute transport with prescribed pore-water transport mechanisms. By matching simulated to measured chlorinity profiles we can infer what the pore-water transport needs to be to explain the measurements.

Pore-water transport model

In our model of solute transport within sediments in the Gippsland Lakes, we assume that salt (NaCl) is a passive solute and that the transport can be described as a diffusive process. The conservation of salt in the sediments can then be written as:

$$\phi \frac{\partial S}{\partial t} = \frac{\partial}{\partial z} \left(\phi D \frac{\partial S}{\partial z} \right) \quad (1)$$

where t is time, z is the vertical coordinate, $S(z, t)$ is salinity, ϕ is sediment porosity, and $D(z)$ is the diffusivity of salt. D would be uniform with depth if diffusion were molecular, but we will investigate another functional form that has diffusivity decreasing with depth within the sediment. Molecular diffusion through water is slow and is further slowed in pore waters because the diffusing molecules must follow a tortuous path around the sediment grains. The diffusivity of the salt in the sediment is estimated by dividing the free-water value by the tortuosity squared (θ^2); that is $D = D_f / \theta^2$. The tortuosity was estimated from the porosity using (Boudreau, 1996):

$$\theta^2 \approx 1 - 2 \ln(\phi) \quad (2)$$

For solution, Eq. 1. requires boundary conditions. At the sediment surface, we assume that the salinity is that measured in the water just above the sediment surface, S_{water} ; that is:

$$S(0, t) = S_{water} \quad (3)$$

At great depth within the sediment, the diffusive transport is set to zero; that is:

$$\left. \frac{\partial S}{\partial z} \right|_{z \rightarrow -\infty} = 0 \quad (4)$$

At the beginning of the simulation, we set the salinity through the sediment column equal to the average salinity over the three-year period of the simulations, \bar{S}_{water} . Thus:

$$S(z, 0) = \bar{S}_{water} \quad (5)$$

The governing equation and the boundary conditions were discretised and then solved using the Crank-Nicolson finite difference method (Roache, 1982).

Porosity

The porosity (ϕ) was calculated from the water content (wc) of the sediment cores (g water per 100 g wet sediment (%)) using

$$\phi = \frac{wc}{(100 - wc)/2.65 + wc} \quad (6)$$

In this equation, the density for the mineral within the sediments has been assumed to be 2.65 g cm^{-3} . The ‘measured’ porosities calculated in this way for all sites are shown in Fig. 2.

For the further analysis of solute transport within the sediments, we developed analytical approximations to the measured porosity profiles. These approximations were obtained by curve fitting using least squares minimisation. The following three analytical functions were tested:

$$\phi(z) = Az^3 + Bz^2 + Cz + D \quad (7a)$$

$$\phi(z) = A + B \tan(Cz + D) \quad (7b)$$

$$\phi(z) = A + B \exp(Cz) \quad (7b)$$

Overall, the best fits were obtained using the third (exponential) functional form. These are the fits shown in Figure 3.

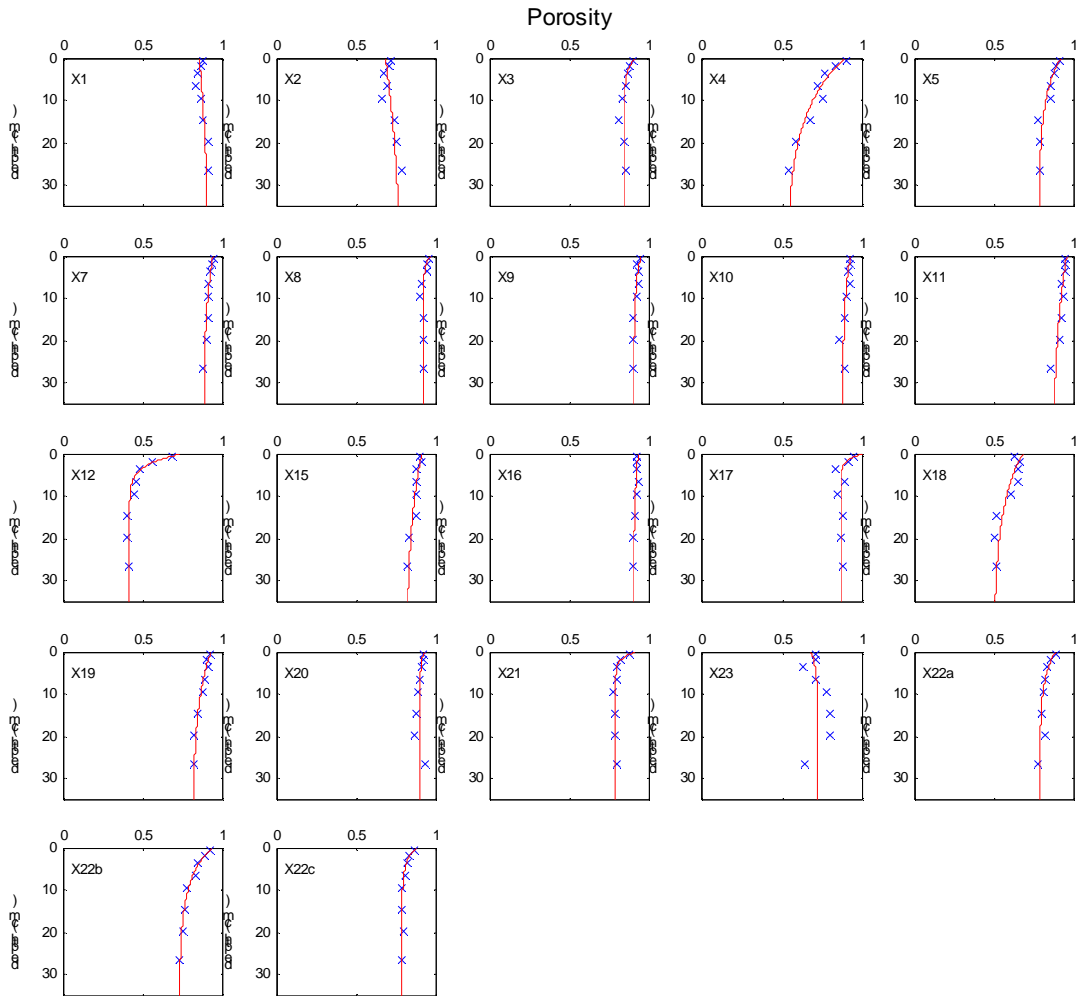


Figure 2. Measured porosities for the cores and the fitted porosity profiles.

Salinity at the sediment/water interface

Water column salinity was measured at 1m increments at 18 sites over the two-year period between 1 July 1997 to 30 June 1999 at weekly intervals by Longmore (2000b). Sampling sites were located throughout Lakes King and Victoria. No measurements were obtained in Lake Wellington as part of this measurement program, so data collected by the Victorian EPA were used instead. The EPA collects salinity data for Lake Wellington (site number 2306) once a month on an ongoing basis at both the surface and bottom of the water column. The salinity in the water above each sediment core (S_{water}) was estimated as the salinity at core depth in the water column monitoring profile at the site closest to the core location. For 1 July 1997 to 30 June 1999, the time series of S_{water} required as the surface boundary condition for the solution of the salinity diffusion equation (Eq. 1) was constructed by interpolating in time the fortnightly estimates of S_{water} obtained from the monitoring program in Lakes King and Victoria. Since the monitoring program in these two Lakes finished in June 1999, we used data from the two-year monitoring program to estimate the seasonal behaviour of S_{water} for the period between 1 July 1999 and the time when the cores were collected (May 2000). In Lake Wellington, S_{water} was estimated by interpolating the monthly salinity measurements at site 2306. Figure 3 shows an example of the time series of salinities obtained at one of the

monitoring stations (site 4) which is close to core site 10 (eastern Lake Victoria) in Fig. 1. The salinity measurements at 9m depth from this monitoring site were used to estimate the time series of S_{water} at core site 10.

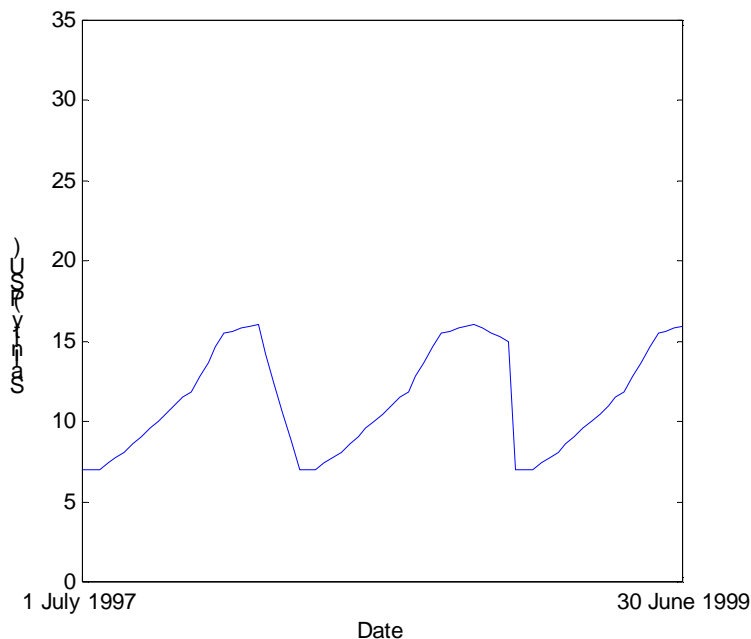


Figure 3. Water column salinity from 1 July 1997 to 30 June 1999 at salinity monitoring site 4.

Inverse approach

In the sediment cores, salinity was assumed equal to chlorinity multiplied by 1.805. For a specified salt diffusivity (D), analytical porosity profile (Eq. 7c), initial condition (Eq. 5), and boundary conditions (Eqs. 3 and 4), Eq. 1 was integrated in time and space to yield a simulated salinity profile for the time of the core measurement. The optimal diffusivity of salt, D , was obtained by fitting the simulated profiles to each measured salinity profile using a Nelder-Mead simplex search algorithm. Two functional forms for D were tested:

$$D = D_0 \quad (8a)$$

$$D = D_m + D_1 \exp(z/10) \quad (8b)$$

In Eqs. 8a and 8b, D_0 and D_1 are fitted constants. In Eq. 8b, $D_m = 11 \times 10^{-6} \text{ cm}^2 \text{ s}^{-1}$ is the diffusivity of NaCl in open water at a temperature of 15 °C a representative average temperature for the Lakes. The form of Eq. 8b is intended to represent enhanced bioirrigation near the sediment surface. The assumed lengthscale of the bioirrigation is 10cm. Boudreau (1998) in his examination of many sediment profiles has suggested that bioturbation within sediments extends to a depth of ~10cm. Also, as will be discussed later, there is evidence that the zone of organic matter degradation within Gippsland Lakes' sediments occurs to about this depth as well. The results are summarised in Table 2 and the fits for the uniform diffusivity shown in Figs. 4.

The values of D_0 obtained by the fitting procedure (Eq. 8a) vary by about a factor of two and have an average of $\bar{D}_0 = 9.1 \times 10^{-6} \text{ cm}^2 \text{ s}^{-1}$ which is about 20% smaller than the molecular diffusivity $D_m = 11 \times 10^{-6} \text{ cm}^2 \text{ s}^{-1}$. The difference between \bar{D}_0 and D_m is not statistically

significant. If the fitted D_0 are treated as a subset of a population of D_0 , then the estimated expected error in \bar{D}_0 is $\pm 1.4 \times 10^{-6} \text{ cm}^2 \text{ s}^{-1}$ which is similar to the calculated deviation of $\pm 2.0 \times 10^{-6} \text{ cm}^2 \text{ s}^{-1}$. Errors in the calculation of the fitted profiles could occur due to errors in the specification of the salinity in the overlying water as well as in the calculation of tortuosity from porosity (Eq. 2).

Table 2. Fitted parameters D_0 and D_1 for the diffusivity representations Eqs. 8a and 8b. For site 22, the values shown are the averages over the three cores 22a, 22b, and 22c.

Site	D_0 $\times 10^{-6} \text{ cm}^2 \text{ s}^{-1}$	r^2	D_1 $\times 10^{-6} \text{ cm}^2 \text{ s}^{-1}$	r^2
1	10.08	0.83	13.0	0.83
2	8.36	0.92	31.4	0.92
3	9.69	0.9	30.4	0.9
4	5.81	0.85	15.1	0.87
5	6.75	0.92	20.7	0.92
7	8.69	0.81	12.9	0.81
8	8.64	0.80	22.2	0.82
9	10.96	0.67	8.8	0.67
10	11.57	0.82	8.3	0.85
11	9.75	0.85	5.0	0.85
12	8.31	0.74	32.2	0.74
15	8.69	0.65	2.1	0.66
16	9.75	0.96	6.6	0.98
17	9.69	0.90	21.4	0.91
18	14.23	0.77	37.7	0.77
19	11.13	0.82	37.2	0.87
20	5.42	0.80	16.6	0.88
21	9.75	0.90	23.5	0.90
22	7.75	0.85	18.8	0.85
23	7.31	0.88	4.9	0.92

As for the analysis using the diffusivity profile obtained from Eq. 8a, the analysis using Eq. 8b shows considerable variability in the value of the fitted parameter D_1 . From the results, the highest values of D_1 are at sites 2, 3, 5, 12, 17, 18, 19, and 21. From the description of the cores in the previous section, these sites all correspond to sites where the water is less than 3 m deep. Also, the surface of the cores from sites 12, 18 and 19 have reports of biological activity. Shallow sites are also likely to be more subject to wave action which can enhance pore-water transport and resuspend bottom sediments (Huettel and Webster, 2000). At the shallow sites it seems likely that there is some enhancement of surficial diffusivities, but this is only of the order of a factor of two or three above molecular diffusion. In any event, the r^2 values for the fits 8a and 8b are not much different from one another so that the assumption that transport was dominated by molecular diffusion is not unreasonable. In further analysis, we shall assume that transport is due to molecular diffusion.

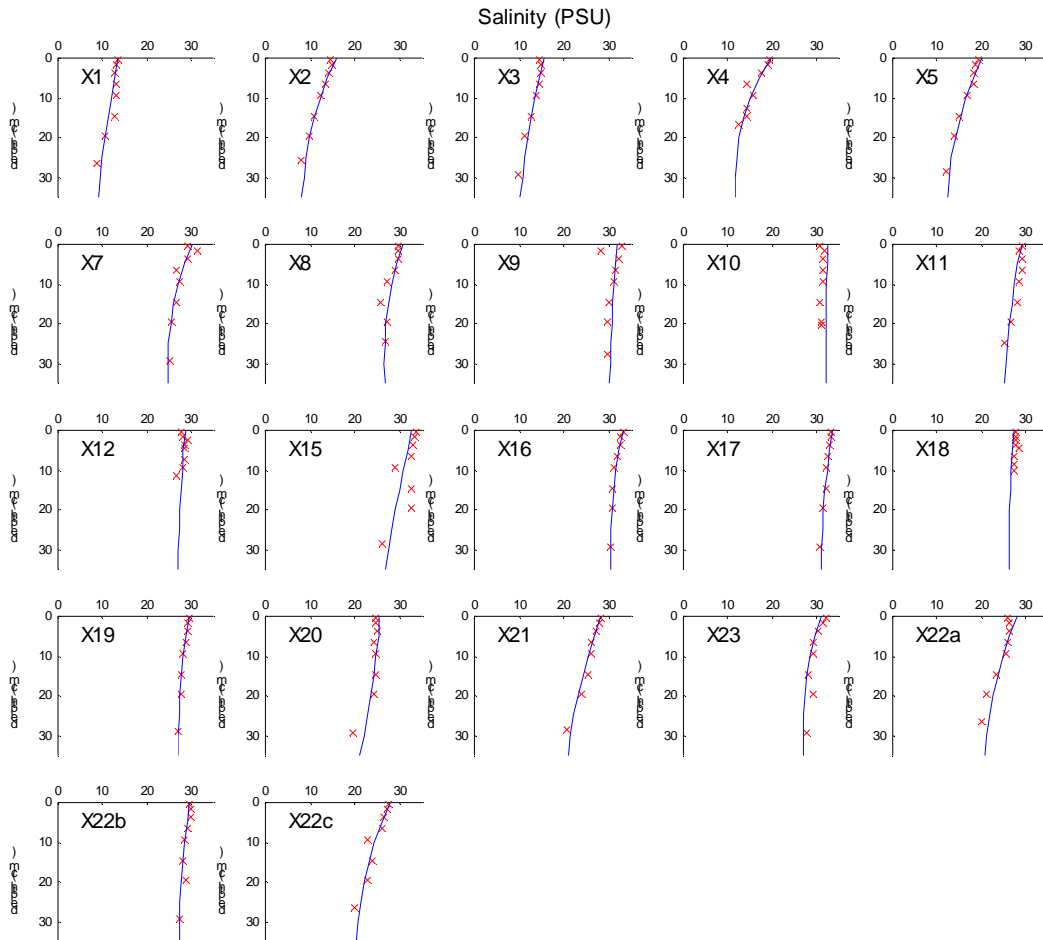


Figure 4. Comparison between measured and fitted salinity profiles within the cores for a uniform diffusivity (Eq. 8a).

Sedimentation Rate

The only study of sediment deposition rates in the Gippsland Lakes is in an unpublished thesis by Reid (1989). Reid estimated the sedimentation rate in Lake Wellington from one sediment core taken in the middle of the Lake. The results give an average, pre-European sedimentation rate in Lake Wellington of 0.3 mm/year and 0.6 mm/year during the post-European period. Based on these figures, the average total deposited mass of sediments in Lake Wellington is 47,000 T/year (assuming a surface area for Lake Wellington is $\sim 148 \text{ km}^2$ and a sediment density of 2.65 g cm^{-3} , porosity $\phi = 0.8$).

A comparison of the sedimentation rates estimated by Reid (1989) from a sediment core was carried out from estimated sediment loads into the lakes over the period July 1997 to July 1999. These loads are based on the analysis by Grayson et al. (2001) which forms part of the GLMP. The average sediment loads for each river into the lakes are as follows: Tambo - 5,000 T/year, Nicholson - 5,000 T/year, Mitchell - 25,000 T/year, Avon - 15,000 T/year, Thomson - 10,000 T/year, LaTrobe - 25,000 T/year. The average load into Lake Wellington (the sum of the Avon, Thomson and LaTrobe) is roughly 50,000 T/year is good agreement with the estimate obtained from calculations based on the data of Reid (1989). Calculating the sedimentation rate for the rest of the Gippsland Lakes from the sediment loads into the lakes over the period July 1997 to July 1999 gives a lake-wide rate of 0.5 mm/year.

The period July 1997 to July 1999 is considered to be a "drier than average" period and as a result could be considered to underestimate the flow and thus sediment delivery into the Gippsland Lakes. Further, there was only one appreciable flood event over the period July 1997 to July 1999 during which most of the sediment is delivered into the Gippsland Lakes. A review of sediment loads into the Gippsland Lakes from sporadic EPA intensive sampling periods is presented below in Table 3 (Robinson, 1995).

Table 3. Estimates of sediment loads from EPA intensive sampling periods (Tonnes).

	77/78	78/79	80/81	84/85	88/89	89/90	Average
Tambo	21,172	1,737	49	874	5,074	4,359	5,544
Nicholson	6,881	509	5	119	n/a	n/a	1,879
Mitchell	48,960	18,829	1,266	16,466	15,363	109,217	35,017
Avon	56,257	1,966	235	1,147	10,358	261,035	55,166
Thomson	96,935	16,286	3,113	7,732	5,567	113,916	40,591
Latrobe	221,226	60,985	38,517	47,014	52,290	64,853	80,814

The data show the large variability in sediment delivery into the Gippsland Lakes. The average sediment loads calculated from the EPA data are roughly a factor of two greater than the sediment loads calculated from the period July 1997 to July 1999. Calculating the sedimentation rate from the average loads from the EPA data gives 2.3 mm/year for Lake Wellington and 1.3 mm/year for the entire Gippsland Lakes. These sedimentation rates could be considered to be an over-estimation because during extreme flood events, a large amount of sediment could potentially be flushed from the Lakes, deposition is not even over the Lakes and a large amount of sediment will be deposited on river deltas and on flood plains.

In summary, a sedimentation rate of 0.6 mm/year calculated on the basis of a sediment core in Lake Wellington can be supported by calculations based on sediment loads. However, 0.6 mm/year should be considered to be a conservative value because of both uncertainties in extrapolating a sedimentation rate based on one sediment core in Lake Wellington to the rest of the Gippsland Lakes, and the large variability in sediment loads to the Gippsland Lakes.

Long-Term Release of Phosphorus and Nitrogen from Sediments

Nutrient fluxes

The diffusive flux of a substance having pore-water concentration C is given by:

$$F = -\phi D_c \frac{\partial C}{\partial z} \quad (9)$$

where D_c is the free-water diffusion coefficient for the solute corrected for tortuosity using Eq. 2. For phosphate and ammonia, we set the free-water diffusivities to be $D_f = 8.46 \times 10^{-6} \text{ cm}^2 \text{ s}^{-1}$ and $D_f = 16.8 \times 10^{-6} \text{ cm}^2 \text{ s}^{-1}$, respectively (Li and Gregory, 1974). The steady-state profile of a solute such as phosphate or ammonia within the sediment can be determined by solving the following conservation equation:

$$-\frac{\partial}{\partial z} \left(\phi D_c \frac{\partial C}{\partial z} \right) = P \quad (10)$$

This equation balances the divergence in the vertical flux of solute against the rate of local production, P . For phosphate and ammonia, this production term would be due to the degradation of buried organic matter. This equation applies in regions of the sediment column where precipitation and adsorption can be neglected. These areas are expected to occur at depths below the top centimetre of the sediment where the sediments would remain anoxic.

When Eq. 9 is integrated over the near-surface sediment layer between $z = -\Delta$ and the sediment surface, $z = 0$, we obtain:

$$-\phi D_c \left. \frac{\partial C}{\partial z} \right|_{z=-\Delta}^{z=0} = \int_{-\Delta}^0 P dz \quad (10)$$

The term on the left hand side of this equation is just the difference in the flux between the sediment surface and the flux at sediment depth Δ .

Figure 5 compares the phosphate fluxes as a function of depth for five groupings of sites. These groups are Lake Wellington, Lake Victoria, South Lake King, North Lake King, and shallow sites. The shallow sites, all of 2.2m depth or less, are located in Reeves Channel (site 14), in South Lake King (site 15) and in Jones Bay (sites 20 and 21). At most sites, the phosphate fluxes show a similar pattern: fluxes are largest at the surface and decline sharply with depth. At 10cm depth, the fluxes are generally only a fraction of those at the surface indicating that most of the production of phosphate occurs between this depth and the sediment surface. The most significant deviation from this general pattern occurred at three of the four shallow sites (sites 12, 14, and 20) where fluxes tended to increase towards the bottom of the profile. At sites 12 and 14, fluxes were negative (downwards) at the surface. Site 12 showed evidence of significant bioturbation activity as did site 11 in South Lake King which is the only other site with a negative surface flux (Table 1). A significant feature of the phosphate fluxes from Lake Wellington is that they become negative at all five sites below about 10cm depth. A possible explanation for this behaviour is that conditions deeper in the cores are conducive to precipitation of phosphate as calcite or some other mineral. In effect, this precipitation would act as a deep sink for phosphate within the sediment column.

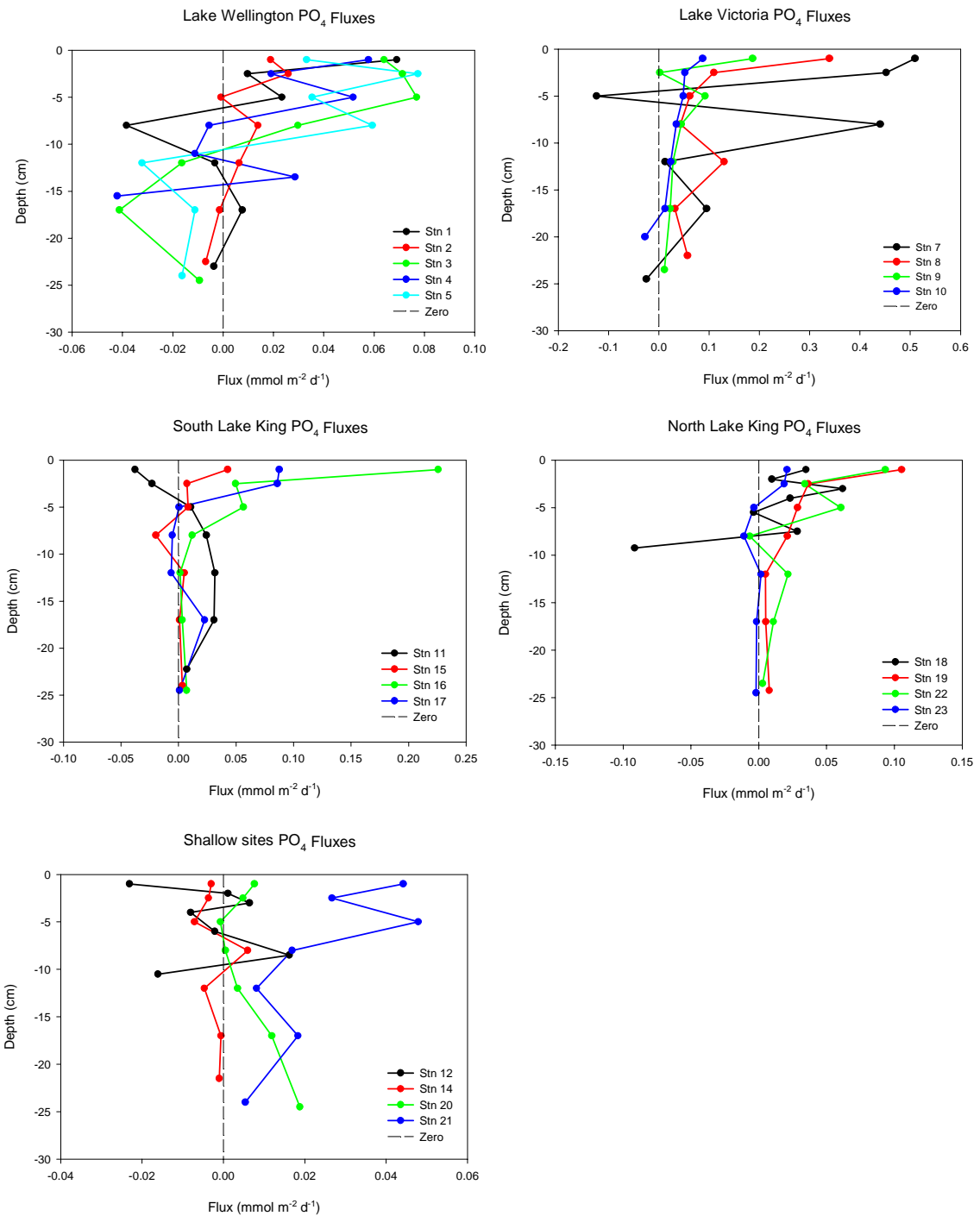


Figure 5. Phosphate fluxes computed from core profiles using Eq. 9.

The ammonia fluxes are shown in Fig. 6 for the same groupings of sites. Ammonia flux profiles showed a generally similar pattern to the phosphate flux profiles. Ammonia fluxes tended to be largest at most sites either at or close to the sediment surface and declined at depth. At these sites, fluxes at $z = -10$ cm were about half those at the surface, whereas for the corresponding phosphate profiles fluxes at this depth were mostly less than a fifth of those near the surface. At a depth of 20 cm though, the ammonia flux had mostly disappeared. It seems that the ammonia flux derives from production over a depth range somewhat greater

than the 10cm depth range of phosphate production. As with phosphate, some of the ammonia flux profiles showed significant deviation from the pattern of decrease from a maximum at the surface to near zero at depth. Sites 1, 11, 19, 20, 21 and 22 show a relatively uniform flux down to a depth of about 10cm and a local maximum at greater depth. We suggest that such a computed flux profile is due to the presence of significant bioirrigation within the sediment column. Certainly, evidence of the presence of benthic organisms is apparent at sites 1, 11, and 19 (Table 1). Sites 20 and 21 in Jones are shallow enough (~2m) that significant biotic activity within the sediments is likely.

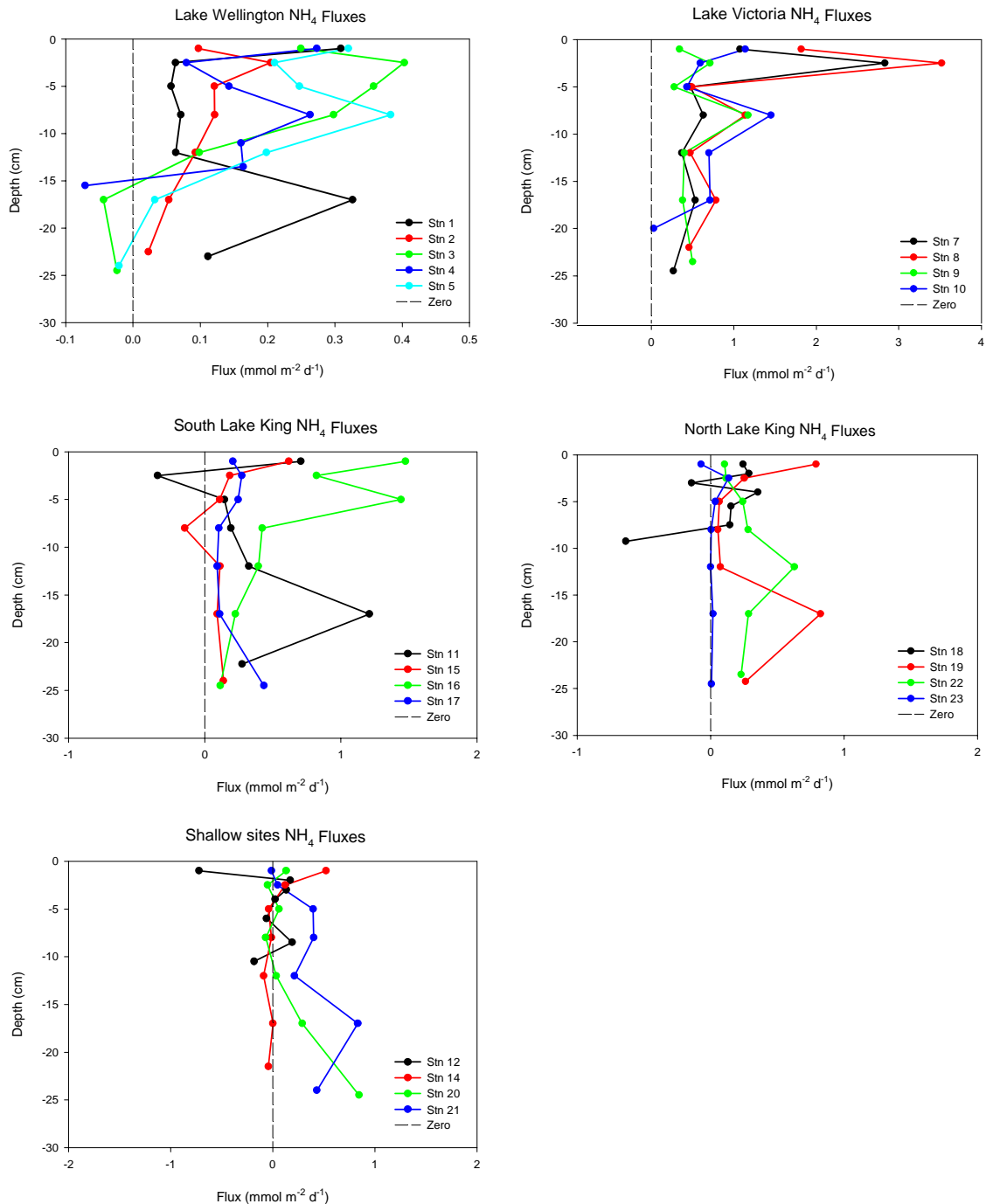


Figure 6. Ammonia fluxes computed from core profiles using Eq. 9.

The effects of bioirrigation on the pore-water profiles from which fluxes are calculated is readily seen by comparing ammonia profiles from sites 11 and 16 (Fig. 7). The profile from

site 16 is the type of profile that we would expect if transport within the sediments were due to molecular diffusion; that is, the magnitude of the vertical gradient of the profile concentrations shows a steady decrease with depth. Conversely, the profile from site 11 shows a surficial zone of approximately 15cm thickness where concentrations are reduced and their gradients are relatively small in magnitude. We suggest that concentrations are reduced within this zone due to enhanced transport caused by bioirrigation (Hancock et al. 2000). In the zone where bioirrigation is significant, Eq. 9 can not be used to calculate the vertical flux. The calculated ammonia fluxes in this zone for sites 1, 11, and 19-22 are likely to be substantially underestimated.

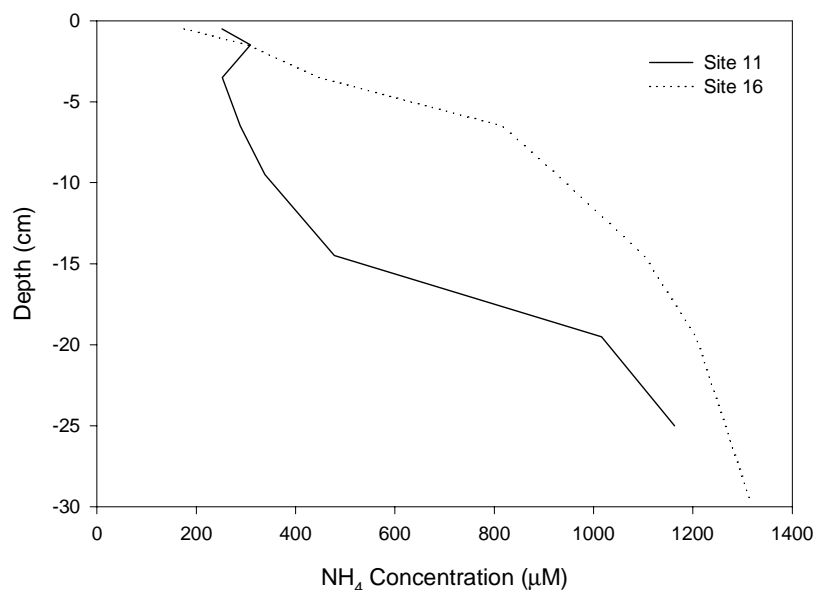


Figure 7. Comparison of ammonia concentration profiles at sites 11 and 16.

The store of labile nutrients within sediments

Notwithstanding the likely underestimation of the fluxes within surficial sediments due to bioirrigation, it seems that most of the production of ammonia and phosphate due to organic matter breakdown within the sediments occurs within 20 cm of the sediment surface. Some of the organic matter breaks down relatively quickly (labile) over time scales of a year or less, whereas other organic matter is essentially inert (refractory) taking hundreds of years if not longer to eventually decompose. We wish to estimate the size of the labile store of organic matter since it is the breakdown of this store which would be primarily responsible for the ongoing release of nutrients from the sediments if the deposition rate of organic matter were reduced.

The maximum store of N and P contained in the labile organic matter can be calculated by integrating the concentrations of solid-phase N and P over the top 20cm of the sediment column. This estimate effectively assumes that all the organic matter is labile which is certainly not the case. Since the upward flux of phosphate and ammonia at 20cm depth is close to zero, it might be assumed that the concentrations of solid-phase N and P at this and greater depths represent the refractory component since all the labile component appears to have been exhausted. Suppose C_{sp} (in $\mu\text{mol g}^{-1}$) is the measured solid-phase concentration of N or P (per unit of sediment mass) and C_{sp}^{20+} is the average solid-phase concentration between a core depth of 20cm and the greatest depth within the core. If C_{sp}^{20+} is taken to be the

concentration of refractory solid-phase nutrient throughout the core then the store of nutrient associated with the labile organic matter is estimated to be:

$$M_{lab} = \int_{-20cm}^0 (C_{sp} - C_{sp}^{20+}) (1 - \phi) \sigma dz \quad (11)$$

Figure 8 exemplifies this method of estimating the store of organic matter using the measured profile of solid-phase N from site 7. The labile matter store is related to the area between the vertical dashed line ($C = C_{sp}^{20+}$) and the solid line ($C = C_{sp}$). In evaluating the integral in Eq. 11, the concentrations of N as mass per mass of sediment solid material is converted to mass per volume.

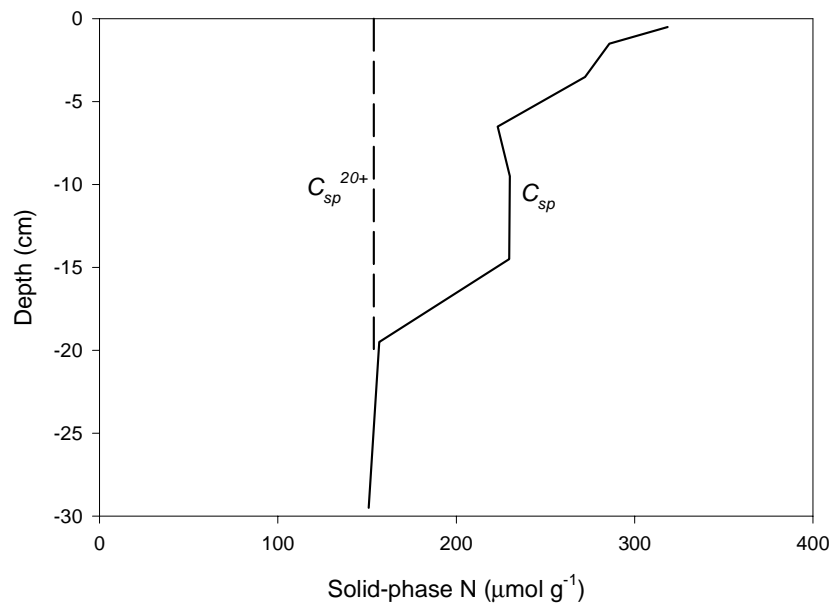


Figure 8. The measured profile of solid-phase N from site 7 showing the estimated concentration of the refractory material, C_{sp}^{20+}

Calculation of time scale for long-term nutrient release

If we assume that the rate of decomposition of the organic matter associated with each nutrient exactly balances the flux of nutrient across the sediment surface, then we can calculate a time for depletion of the nutrient store within sediments assuming that there is no resupply. This time scale is:

$$T = \frac{M_{lab}}{F_0} \quad (12)$$

where M_{lab} is the mass of the nutrient in the labile store and F_0 is the flux of nutrient across the sediment surface evaluated using Eq. 9. Table 4 presents calculated values of T for the cases of N and P stores estimated using Eq. 11 (T_N and T_P). Values of T are also presented for stores calculated with the assumption that all solid-phase nutrient between 0 and 20cm depth is labile

(T_{N-all} and T_{P-all}). In effect, the latter calculations assume that $C_{sp}^{20+} = 0$ in Eq. 11. Those measured profiles which did not extend to sufficient depth are not included in the Table. The

missing values in the Table refer to sites for which either the calculated store was negative (profile increasing with depth) or for which the surface flux was negative.

Table 4. Calculated time scales for labile matter depletion within sediments. Sites at which dissolved nutrient profiles were noticeably perturbed by bioirrigation are marked with an asterisk.

Site	T_P (y)	T_N (y)	T_{P-all} (y)	T_{N-all} (y)
1*	3	-	38	178
2	5	-	176	320
3	16	-	50	139
5	23	4	104	100
7	1	9	5	28
8	1	3	6	18
9	1	17	10	87
10	-	4	24	30
11*	-	28	-	76
14	-	-	-	58
15	14	23	63	85
16	2	6	10	28
17	1	4	31	232
19*	7	23	29	66
20*	52	-	278	575
21*	17	-	103	-
22*	7	61	43	505
23	18	-	160	-

There is a high degree of variability in the calculated time scales between sites. For T_P and T_N , this is due to variability in both the store size and in the surface flux. For T_{P-all} and T_{N-all} , the store sizes were mostly within a factor of two of one another between sites, so that their variability was mainly due to the surface fluxes. In most cases, time scales for N depletion are considerably longer than those for P. The calculated surface fluxes at the sites with significant bioirrigation may be underestimated substantially which would cause the calculated depletion time to be too large by an unknown amount. Site 20 showed the largest values of T_P , T_{P-all} and T_{N-all} of any of the sites and this is a site that also shows evidence of bioirrigation activity in the solute profiles. If the sites showing bioirrigation are excluded, then the average time scales for Lakes Victoria and King are $\bar{T}_P = 5$ y, $\bar{T}_N = 9$ y, $\bar{T}_{P-all} = 39$ y, and $\bar{T}_{N-all} = 71$ y. When only a fraction of the solid-phase nutrient is assumed to be labile, the 'clearance' time scale for nutrients within the sediments is of the order of ten years. In the worst case scenario, which assumes all the solid-phase nutrient in the top 20cm of the sediment to be labile, the clearance time scales increase by a factor of about eight for both P and N. In Lake Wellington, T_N could be calculated for site 5 only. For all the other sites in the Lake, the concentrations of solid-phase N increased with depth. Excluding site 1 which shows significant bioirrigation, the average depletion time scales are $\bar{T}_P = 8$ y, $\bar{T}_{P-all} = 110$ y, and $\bar{T}_{N-all} = 186$ y which are significantly larger than those calculated for Lakes Victoria and King.

Depending on which assumption is made about the degradability of the solid-phase N and P, the time taken for the sediment store to deplete itself ranges from relatively short (~10 y) to

quite long (~100 y). A key assumption required for the analysis leading to the shorter time scale is that the rate of accretion of solid-phase N and P within the sediment column is constant so that the decline in concentration with depth can be attributed to decomposition. It is certain that significant increases in the loads of sediment, nutrients, and organic matter have occurred over the last 100y which could lead to increases in the concentration of buried solid-phase nutrients. Also, the transition of the Lakes from a freshwater system to an estuarine one which occurred with the dredging of the Entrance in 1889 would likely have changed the biochemical and ecological function of the Lakes leading to a change in the properties of the buried organic matter. With a sedimentation rate of ~1 mm/y, we expect that the changes in loads and in exchange through the Entrance that occurred over the last century would affect sediments to a depth of ~10cm. With bioturbation, the depth of influence would have extended to a greater depth. In other words, it is not certain what fraction of the decrease in the concentration of solid-phase nutrients with depth is due to changes in the organic matter being deposited and what fraction is due to the decomposition of its labile component. If the deposition rates of labile solid-phase nutrients have increased significantly over the last century, then T_p and T_N are overestimated. Similarly, T_p and T_N are underestimated if the deposition rates have decreased over this time.

Comparison with fluxes measured by benthic chambers

Benthic chambers were used to measure nutrient fluxes at three sites in eastern Lake Victoria and Lake King by Longmore (2000b). Measurements were obtained on eight occasions between August 1997 and June 1999. At each site, four benthic chambers were installed; the two chambers with transparent lids measured nutrient fluxes under illuminated conditions and the two chambers with opaque lids measured release under dark conditions. To obtain fluxes representative of averages over the diurnal cycle we have averaged the fluxes measured by the transparent and opaque chambers. Figures 9 and 10 compare these fluxes also averaged over the eight sampling times to the fluxes calculated from the sediment cores using Eq. 9.

Except for the site in Lake Victoria, the benthic chambers show phosphate fluxes that are several times larger than those calculated from the cores collected at nearby sites. In Lake Victoria one of cores (site 8) showed a phosphate flux that was only about 15% less than the nearby benthic chamber measurement. All the ammonia fluxes estimated from the cores were of the order of half or less than those obtained from the benthic chambers. The likely cause of the discrepancy is that the cores do not provide a measurement of the true flux across the sediment surface. The 'surface' flux is calculated from the difference in nutrient concentration between depths of 0.5 and 1.5cm; that is, the flux pertains to an effective depth of 1cm. This flux would not include the contribution to the flux from the decomposition of highly labile organic material decomposing on the sediment surface, whereas the benthic chambers would. At times of net growth, microphytobenthos living on the sediment surface could even reduce surface fluxes. Figure 11 shows the fluxes of phosphate and ammonia from the benthic chambers averaged over the three sites for each sampling time. The figure demonstrates that the surface fluxes vary considerably over the two-year sampling period. Longmore (2000b) relates peaks in benthic fluxes to blooms of phytoplankton in the water column. These are times when the concentrations of labile organic matter on the sediment surface would be expected to be high.

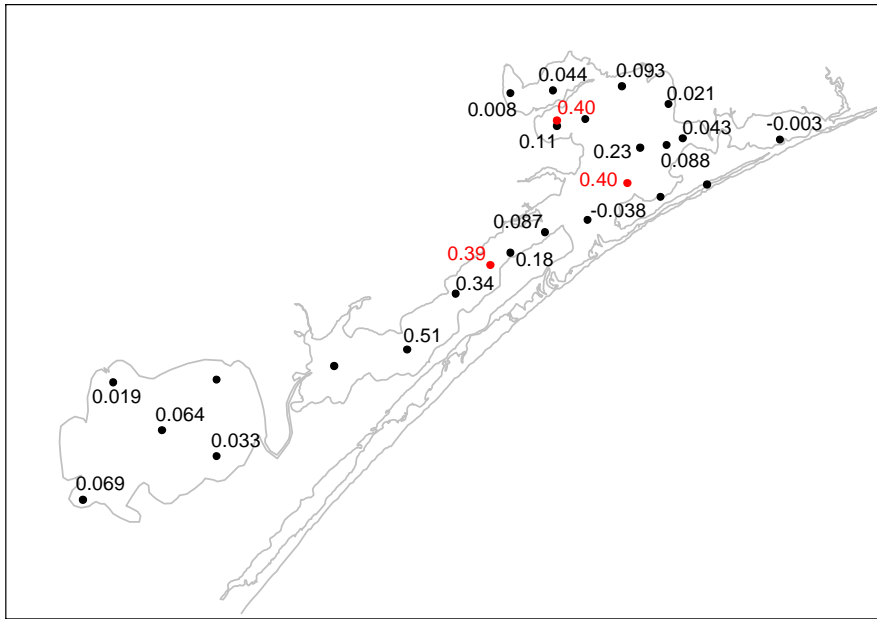


Figure 9. Surface phosphate fluxes from core measurements (black) and from the benthic chambers (red). All units are mmol m⁻² d⁻¹.

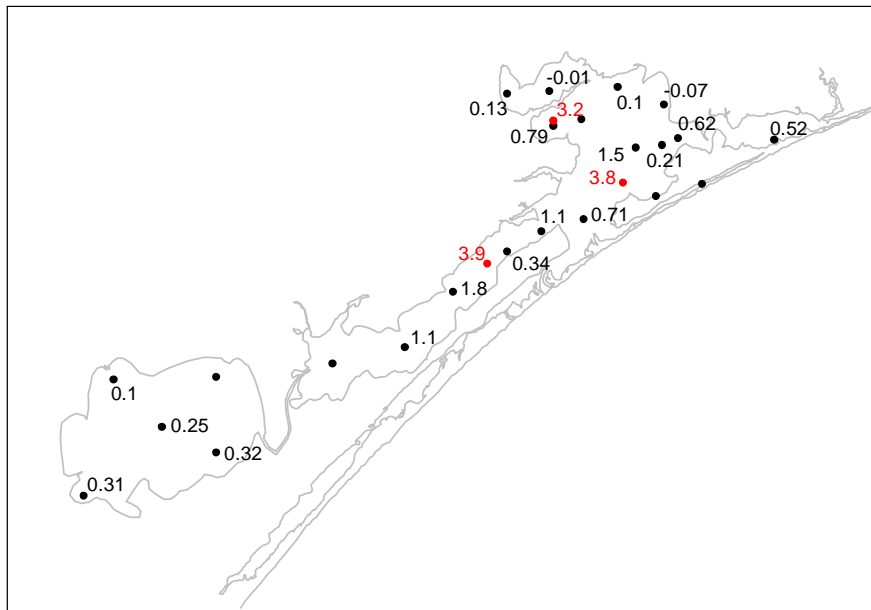


Figure 10. Surface phosphate fluxes from core measurements (black) and from the benthic chambers (red). All units are mmol m⁻² d⁻¹.

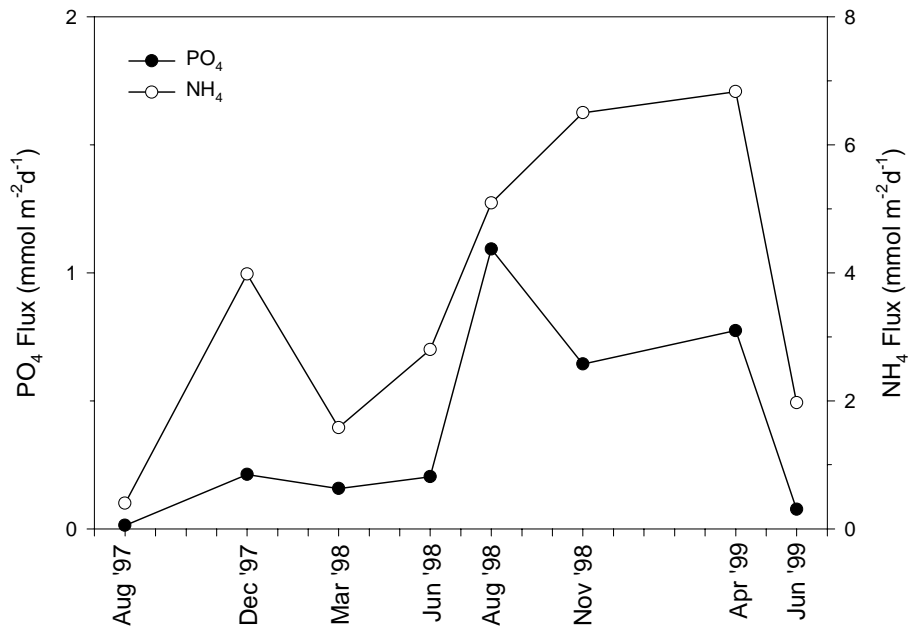


Figure 11. Time series of benthic chamber fluxes of phosphate and ammonia averaged over three sites.

The fluxes measured by the benthic chambers and those calculated from the nutrient concentrations in the cores are not inconsistent with one another. We consider that it is the fluxes calculated from the core measurements that are most pertinent to the estimation of long-term nutrient release from the sediment column. It is these fluxes which represent the removal of nutrients from the sediment store below the top few millimeters of sediment.

Summary

1. At most sites that were sampled throughout the Lakes transport within the sediment pores appears to be dominated by molecular diffusion. At some sites there was clear evidence of bioirrigation that would increase the exchange of pore water through surficial sediments. Significant bioirrigation is more common and should be expected in the shallower regions of the Lakes.
2. The sedimentation rate in Lake Wellington has been measured directly to be 0.6mm/y, but this estimate is based on a single core. On the basis of sediment loads to the Lakes and assuming 100% retention, sedimentation rates of 2.3mm/y is calculated for Lake Wellington and 1.3mm/y is calculated for the other Lakes. These numbers are certain to be overestimates – a sedimentation rate of $\sim 1.0 \pm 0.5$ mm/y for the Lakes is a likely range.
3. The estimated time taken for the sediment store to deplete itself after it stops receiving organic matter from the water column ranges from relatively short (~ 10 y) to quite long (~ 100 y). This time range depends on which assumption is made about the degradability of the solid-phase N and P within the sediments. Another key assumption is that transport is dominated by molecular diffusion near the sediment surface which we know to be untrue at some sites. It is likely though that the ‘long’

time scale is a substantial overestimate of the true depletion time. Uncertainties in the depletion time could be reduced with measurements of solute transport rates, sedimentation rates, and organic matter 'age'.

4. In this analysis, the times for the clearance of labile nutrients from 'deep' sediments do not consider possible replenishment from the water column. Also, the likely significance of deep sediment release for the overall cycling of nutrients in the Lakes needs to be considered. Both these issues involve water column and surficial sediment processes and will be addressed in conjunction with results from the integrated biochemical/ecological model that has been developed for the Lakes.

References

Boudreau B. P. (1996) The diffusive tortuosity of fine-grained unlithified sediments. *Geochim. Cosmochim. Acta* **60**, 3139-3142.

Boudreau, B.P. (1998) Mean mixed depth of sediments: The wherefore and the why. *Limnol. Oceanogr.*, **43(3)**, 524-526.

Grayson, R., Tan, K.-S. and Western, A. (2001) Estimation of sediment and nutrient loads into the Gippsland Lakes. CEAH Report No 2/01.

Hancock, G.J., Webster, I.T., Ford, P.W., Moore, W.S. (2000) Using Ra isotopes to examine transport processes controlling benthic fluxes into a shallow estuarine lagoon. *Geochim. Cosmochim. Acta* **64(21)**, 3685-3699.

Li Y.-H., and Gregory S. (1974) Diffusion of ions in sea water and in deep-sea sediments. *Geochim. Cosmochim. Acta* **38**, 703 – 714

Longmore, A.R. (2000a). Gippsland Lakes sediment nutrient inventory. Marine and Freshwater Resources Institute Report No. 19. Fisheries Victoria, Victoria Australia.

Longmore, A.R. (2000b). Effects of water quality on black bream habitat in the Gippsland Lakes. Marine and Freshwater Resources Institute Report No. 42. Fisheries Victoria, Victoria Australia.

Reid, M. (1989) Paleo-ecological changes at Lake Wellington, Gippsland Lakes, Victoria during the late holocene: A study of the development of a coastal lake ecosystem. Unpubl. BSc. Honours thesis, Department of Geography and Environmental Science, Monash University.

Roache, P.J. (1982) Computational Fluid Mechanics. Hermosa, Albuquerque, N.M.

Robinson, D.P. (1995) Water quality and nutrient status of Lake Wellington. EPA Victoria Publication 460.

Isotherm Analysis of Sediment Samples and Water Column Samples Collected in the Gippsland Lakes – 2000 - 2001

Ian T. Webster, CSIRO Land & Water
Michael Grace, Water Studies Centre, Monash University

July 2001

Introduction

Phosphorus is a highly particle reactive element and the inclusion of the dynamics of its interaction with both suspended and benthic sediments in the biogeochemical model of the Gippsland Lakes was deemed to be essential. In order to obtain information on the adsorptive properties of P to the Lakes' sediments, the Gippsland Lakes Particle Sizing and P Sorption Isotherm task was commissioned from the Water Studies Centre at Monash University. Dr. Mike Grace from the Water Studies Centre led this project. The project is now complete and its results have been made available through the report 'Short Progress Report on P Sorption & Particle Sizing (March 2001)' by Drs Mike Grace and Soheyl Tadjiki and through a series of spreadsheets. The report is attached as an Appendix to this report.

This report is intended to provide a further analysis of the data provided by Drs. Grace and Tadjiki. Specifically, the analysis is intended to provide the information on P adsorption behaviour necessary for the implementation of the biogeochemical model to the Gippsland Lakes as part of the Gippsland Lakes Environmental Study. The following report considers first the adsorption behaviour of benthic sediment samples collected throughout the Lakes. Next is an analysis of the suspended sediments collected from Lake Wellington. Finally, the Conclusions are presented.

Benthic Sample Analyses

Benthic sediment samples were collected from 24 sites throughout the Gippsland Lakes on January 3 and 4, 2001. The samples were collected by P. Ford, R. Grayson and I. Webster using a van Veen grab. Figure 1 shows the locations of the sampling sites; Table 1 summarises features of these sites. The sediments were divided into groups according to their locations, depths and physical appearance. These groups are given the identifications A to H and their broad characteristics are outlined in Table 2.

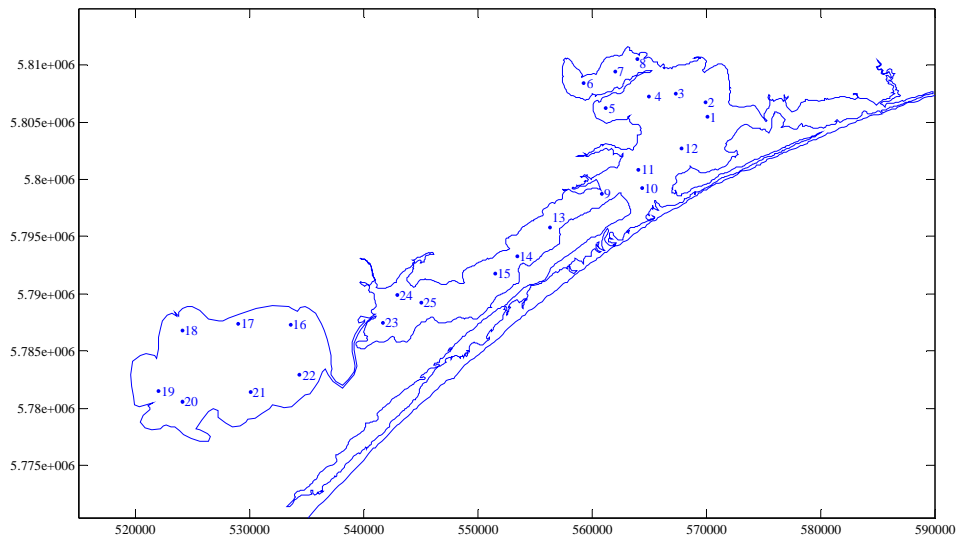


Figure 1. Locations of sediment sampling sites January 3-4, 2001.

Table 1. Summary of sites for sediment sample collection.

Site	S (‰)	Depth (m)	Location	Group
1	36.98	6.3	N Lake King	G
2	35.58	6.3	"	G
3	36.37	7.1	"	G
4	31.78	6.7	"	G
5	29.5	4.1	"	G
6	14.5	1.4	Jones Bay	A
7	22.12	2.1	"	A
8	24.97	2.1	"	A
9	40.64	9.8	S Lake King	F
10	29.13	2.2	"	E
11	40.06	8.4	"	F
12	35.25	7.2	"	G
13	15.31	6.5	E. Lake Victoria	E
14	19.03	5.2	"	E
15	25.37	5.1	"	E
16	6.51	3.0	Lake Wellington	D
17	9.06	3.0	"	D
18	8.43	2.5	"	C
20	7.5	2.6	"	C
21	9.76	3.1	"	D
22	12.45	2.9	"	B
23	22.51	1.7	W. Lake Victoria	H
24	25.62	2.3	"	H
25	27.53	3.1	"	H

Table 2. Summary for sediment samples showing groupings of sites for subsequent analysis.

Group	Location	Depths (m)	Common characteristics
A	Jones Bay	1.4-2.1	Thin oxic layer, benthic weed
B	SE Lake Well.	2.9	Only sandy sediment in Lake Wellington
C	W. Lake Well.	2.5-2.6	2cm thick brown surface layer
D	E. Lake Well.	3.0-3.1	"
E	E. Lake Vic.	2.2-6.5	Thin oxic layer, viable flora and fauna at 1 site
F	S. Lake King	8.4-9.8	Deepest sites, anoxic bottom water
G	N. Lake King	4.1-7.2	Very thin oxic layers on sediments
H	W. Lake Vic.	1.7-3.1	5mm oxic layer, evidence of benthic fauna

Isotherm analysis on benthic samples

Benthic sediments from the 24 sites were combined into the eight groups identified in Tables 1 and 2 for analysis. The adsorption properties of the sediments were measured by the Water Studies Centre at Monash University. The analysis method involves adding increasing amounts of phosphate to diluted suspensions of the sediment samples. As the P concentration in the liquid phase increases, so does the amount of P adsorbed to the particles. The partition between the dissolved and adsorbed P phases is inferred by using the radioactive tracer ^{33}P , a small amount (picomolar quantity) of which is added to the samples. The measurement of the numbers of beta particles emitted from ^{33}P on the total sample and in the supernatant allows a direct estimation of the relative concentrations of exchangeable P associated with the solid and liquid phases in each sample. With the relative amounts of P adsorbed and dissolved determined using the ^{33}P analysis and the dissolved P concentration determined using FIA analysis on the supernatant, the absolute concentration of P adsorbed to the particles was calculated. Throughout this work, filtration through a 0.2 μm polycarbonate membrane filter was used as the arbiter between ‘dissolved’ and solid phases.

In all of the analyses, pH was not controlled except for sample C which was subdivided into two subsamples. The adsorption behaviour was determined on these two subsamples separately in order to investigate the effects of pH variation. For one subsample (C7), the adsorption behaviour was determined at a pH of 7.1 and for the other (C8) the pH was 8. The pH was adjusted to the desired pH at the commencement of the experiment. The natural *in situ* buffering of the bicarbonate/carbonate system was used to maintain this pH. All the sediment adsorption experiments were performed after ‘incubation’ periods of 2 and 48 hours. It might be expected that adsorption to surface adsorption sites on the sediment particles to be complete after 2 hours (‘fast adsorption’), but adsorption is expected to be a process that can take days or possibly much longer to approach equilibrium. The slow adsorption is thought to most probably involve the diffusion of adsorbed ions from the particle surfaces to the interiors of the adsorbing layers coating the particles. The adsorption behaviour of the sediment samples was determined over a range of dissolved P concentrations from close to zero up to about $10000\mu\text{gL}^{-1}$. Further analysis is restricted to the concentration range $0\text{-}1000\mu\text{gL}^{-1}$ since this is the range that is most likely to be encountered in the water column and in the surficial sediments throughout the Gippsland Lakes (Longmore, 2000).

Isotherm Fitting Procedure

Figure 2 shows the measured concentrations of adsorbed P expressed as mass of adsorbed P per mass of sediment as a function of the concentration of dissolved P for both the 2 h and for the 48 h incubations. Also shown are the fitted Freundlich isotherms, which have the functional form:

$$P_s = AP_d^B. \quad (1)$$

A and B were estimated from the data using a two-stage fitting procedure. All calculated coefficients assume that the units of P_s are $\mu\text{g g}^{-1}$ and of P_d are $\mu\text{g L}^{-1}$. The log-transform of Eq. 1 is:

$$\log(P_s) = \log(A) + B \log(P_d) \quad (2)$$

In the first step of the fitting procedure, the data were log transformed and fitted to Eq. 2 as a least squares linear regression to determine optimal values of A and B for each sediment sample. The exponents B for the 2-h incubations ranged between 0.35-0.54 and had an average $\bar{B} = 0.45$. For the 48-h incubation, the exponents were significantly smaller ranging

between 0.28-0.45 with an average of $\bar{B} = 0.34$. Golterman (1995) has shown that the Freundlich isotherm provided a very good description of his adsorption data with $B = 0.333$ particularly for $\text{pH} \geq 8$. His incubations were 3-5 d.

In the second step of the fitting procedure, the exponent B was specified as the average determined in the first step (for each incubation period) and A was then estimated for each adsorption experiment using a second least squares fitting procedure also using log-transformed data. The advantage of using log-transforms in the least squares fitting is that neither the low or high concentration ends of the measurements are given undue weight in the calculations. For all the fits, $r^2 > 0.94$ and most had $r^2 > 0.98$. The Freundlich isotherms are a good representation of the data for both incubations even though for each set of incubations the exponent has a single prescribed value.

A more common representation of the adsorption behaviour is the Langmuir isotherm, which is expressed as:

$$P_s = \frac{P_{\max} k P_d}{1 + k P_d} \quad (3)$$

where P_{\max} is the maximum concentration that can be adsorbed and k is the 'Langmuir' constant - a measure of the binding affinity of the substrate. Figure 3 compares the fitted Langmuir isotherms to the Freundlich isotherms for the 48-h incubations. The Langmuir representation is inferior to the Freundlich representation in every case, the average value of r^2 being 0.90 for the former versus 0.99 for the latter.

Freundlich Isotherms - Benthic Sediment

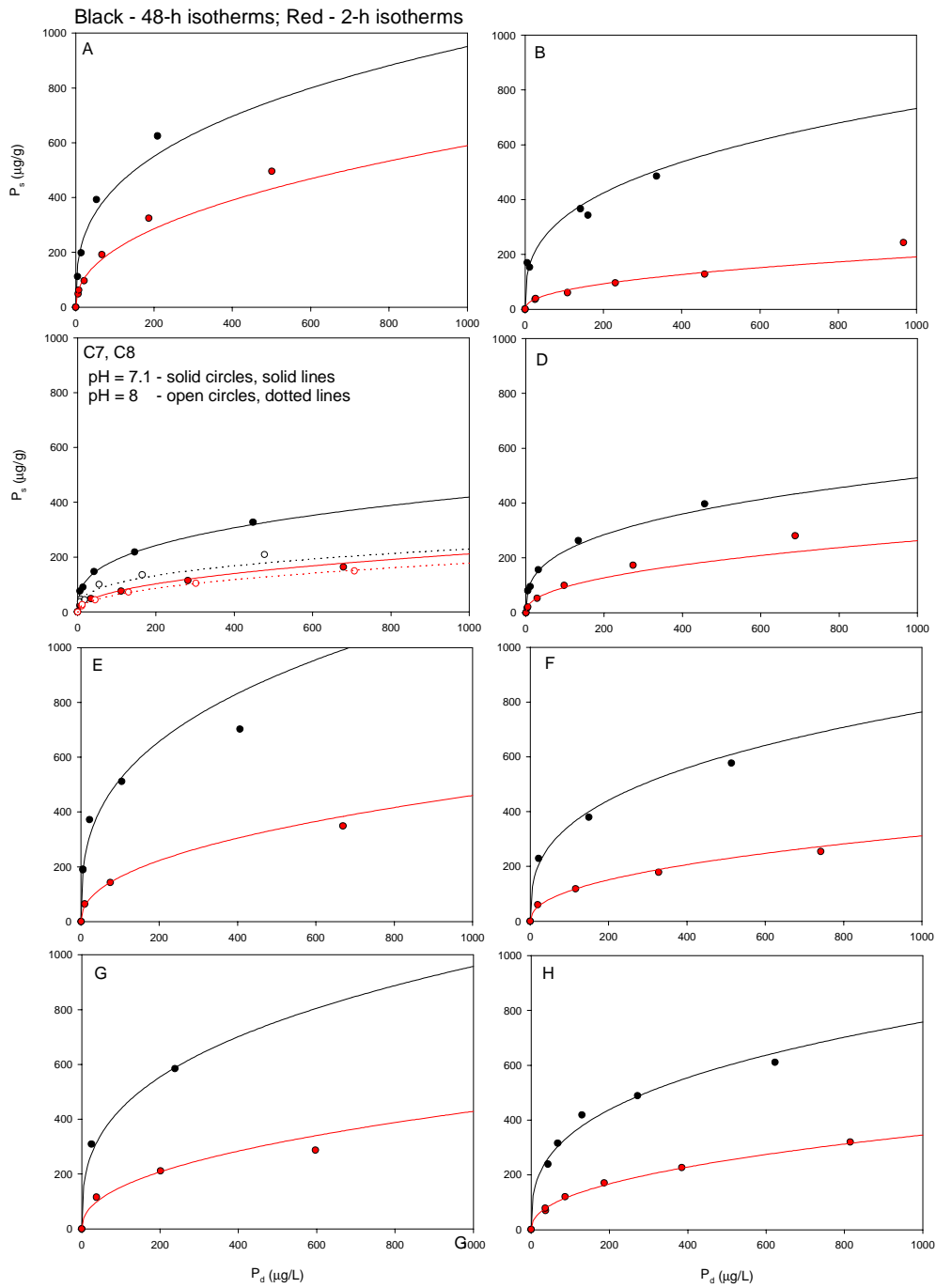


Figure 2. Measurements and fitted Freundlich isotherms for 2-h and 48-h incubations on benthic sediments.

Comparison Freundlich and Langmuir Isotherms - 48 h

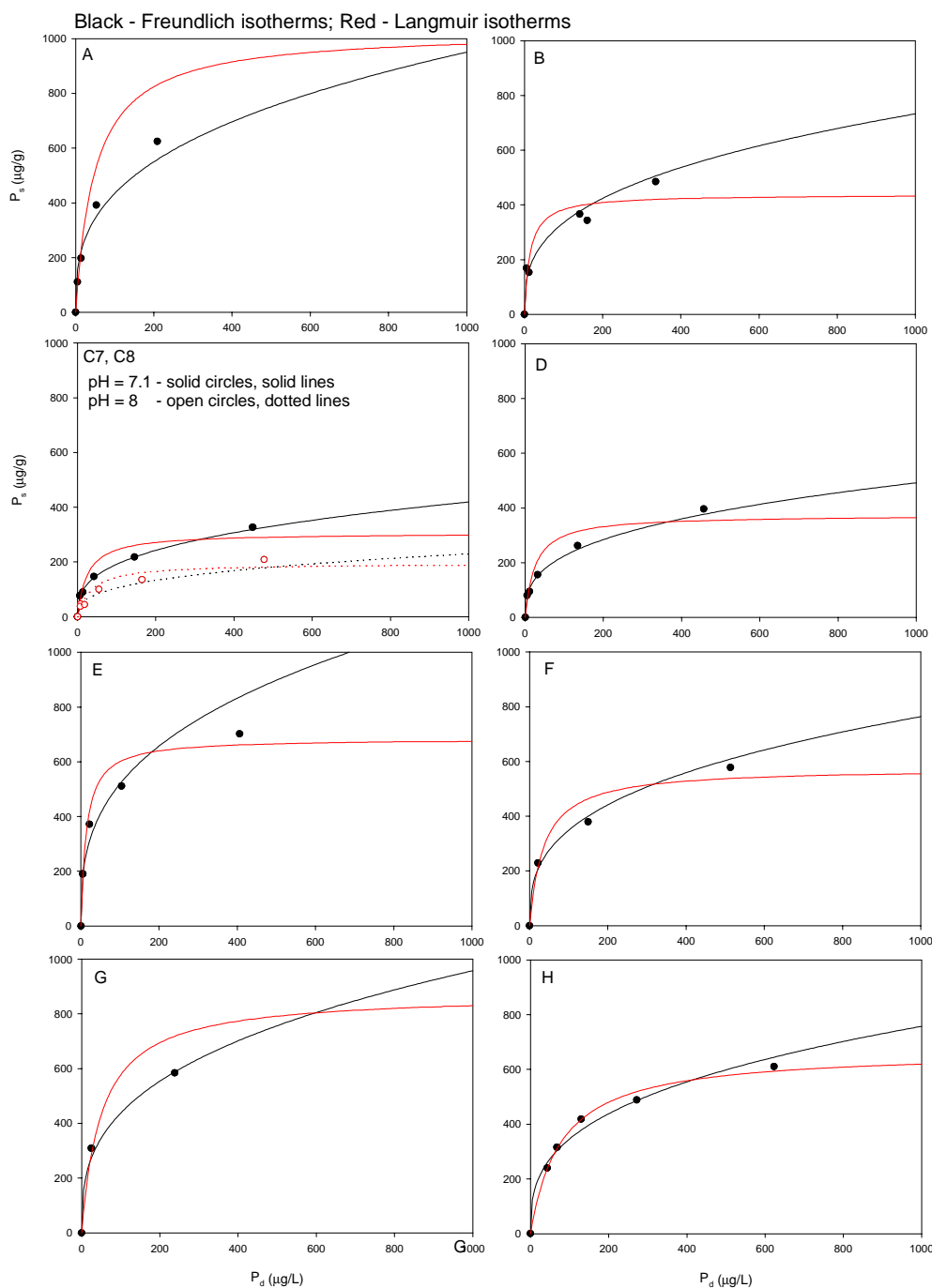


Figure 3. Comparison of Freundlich and Langmuir isotherms for 48-h incubation.

Figure 2 shows that at all dissolved P concentration, adsorption after the 48-h incubation is substantially higher than for the corresponding 2-h incubation. About twice as much P is adsorbed after 48 h as after 2 h at $P_d = 500 \mu\text{g L}^{-1}$. The increased adsorption after 48-h could be due to bacterial uptake during the extra period of incubation or be due to the migration of surface bound phosphate ions into the interiors of the oxide coatings on the particles. The

latter explanation seems to be the more likely due to the regularity in the development of all the isotherms during the longer period of incubation.

Portielje and Lijklema (1993) have undertaken modelling of the adsorption kinetics for particles coated with aluminum oxyhydroxides primarily. They consider a two-day incubation to be 'short-term' and show through models and experiment that adsorption continues to beyond 100 days. The maximum adsorption capacity is shown to be about three times higher than that suggested for a Langmuir isotherm determined after a 2-day incubation. However, this maximum adsorption capacity was estimated in an experiment involving the co-precipitation of Al and Fe oxyhydroxides. Portielje and Lijklema suggested that the timescale for intra-particle equilibration of P depends on the thickness of the oxide coatings.

Experiments by House et al. (1995) show that in suspended sediments P was adsorbed rapidly in the first hour or so and slowed thereafter. At a time of 2 h, the adsorbed concentration was about $5.1 \mu\text{g g}^{-1}$, at 24 h it was $7.1 \mu\text{g g}^{-1}$, and at 48 h at the experiment's end, the adsorbed concentration had increased to about $7.7 \mu\text{g g}^{-1}$. Unfortunately, it is not known how much adsorption would have increased after 48 h.

Also, House et al. suggest that P uptake by bacteria could become significant relative to adsorption after 12 h, but their calculation was not based on any data from their experiment.

Table 3 presents the values of A calculated from the fitting procedure for the 48-h incubation. The adsorptive capacities of the sediments vary by less than a factor of 3 if C8 is excluded. Sample C8 was analysed at a pH significantly higher than its natural pH. The relatively modest variation in adsorptive capacity seems at first glance to be surprising considering that some samples were collected in well oxidised sediments (e.g. Lake Wellington samples C and D) whereas some come from areas often overlain by anoxic bottom waters (e.g. Lake Victoria sample F). However, it is apparent that the concentrations of Easily Extractable Iron are all fairly uniform in the Lakes' sediments (Table 3). The Easily Extractable Iron is determined using a relatively mild extraction method (hydroxylamine and acetic acid) to just extract the amorphous Fe oxyhydroxide surface coatings on the particles. Harsher extractions result in higher Fe contents and would dissolve some of the more crystalline phases that are probably far less important in binding P.

Since the adsorption experiments were performed under oxic conditions, one might assume that the Easily Extractable Iron was present as iron oxyhydroxide coatings on the particles primarily responsible for adsorption. In other words, the isotherms are as uniform as they are because of the uniformity of the concentrations of iron oxyhydroxide. Why should the Fe concentration be so uniform in the sediments? An explanation is that Fe is transported through the water column and from Lake to Lake on suspended particles. When the particles settle to the bottom under oxic conditions, the iron oxyhydroxides on their surface are insoluble and so immobile. Under anoxic conditions the Fe is immobilised as insoluble iron sulfide. Thus, as long as particles sedimenting to the bottom of the Lakes have an approximately uniform Fe concentration, so will the benthic sediments.

Although for both incubation periods the isotherms obtained on sample C showed that pH reduces the adsorptive capacity (Fig. 2), the degree of reduction differs markedly. For the 2-h isotherm, the fitted value of A for a pH of 7.1 (C7) is 1.2 times higher than for a pH of 8 (C8); for the 48-h isotherm, the ratio is 1.8. For a given sediment, it would be expected that the sorption would decrease with increasing pH. Electrostatic repulsion between the negative phosphate ions (HPO_4^{2-} , H_2PO_4^-) and the surface of the particle increases as the organic functional groups on the particle surface become progressively more deprotonated (and hence negatively charged). However, examination of Table 3 suggests that the variation in adsorptive capacity of the sediments is not linked to variation in pH in an obvious way.

Table 3. Values of the fitted constant A for the Freundlich 48-h isotherms, the concentrations of Easily Extractable Iron, and pH at the time of measurement.

Sample	A (48-h)	E. Extr. Fe mg g ⁻¹	pH
A	91	16.4	7.5
B	70	13.0	7.8
C7	40	13.9	7.1
C8	22	12.4	8
D	47	12.9	7.65
E	109	15.0	7.85
F	73	14.5	7.9
G	91	13.8	7.8
H	72	18.5	7.9

It has been hypothesised that the crystalline structure of the Fe (oxides) responsible for P adsorption affect the rate of the slow adsorption (Slomp et al. 1998). Sediments throughout the Lakes do have differing exposures to reducing conditions. Therefore, it seems likely that they would also have different crystal structures in their Fe oxide coatings, and so variations in their propensity to adsorb P.

Suspended Sediment Analyses

Water samples were collected at four sites on three occasions by N. Biggins and A. Stevens of the Victorian EPA for the analysis of the adsorptive properties of suspended sediments. These sites, designated 1-4 in Table 4, were sampled on June 5, August 2, and November 26, 2000. The three sites in Lake Wellington were spaced approximately equidistantly across the width of the Lake. In addition, two water column samples for the adsorption analysis of suspended sediments were collected in Lake Wellington at sites 19 and 22 by Ford, Grayson and Webster on January 4, 2001 during their sampling of benthic sediments. Site 19 was located near where the Latrobe River enters the Lake and site 22 was located near the entrance to McLennans Strait (Fig. 1). Note that EPA sites 1-4 do not correspond to the sampling sites shown in Fig. 1. Also presented in Table 4 are the salinities of the water samples collected. The gradual increase of salinity between sites 3 and 1 reflects increasing distance of the sampling sites away from the discharge from the Latrobe River.

Table 4. Summary of sites for water sample collection.

Site	Location	Salinity (‰) June 5, 2000	Salinity (‰) Aug. 2, 2000	Salinity (‰) Nov. 26, 2000	Salinity (‰) Jan. 4, 2001
1	Lake Well.	15.5	14.2	6.9	
2	"	15.2	13.4	6.2	
3	"	15.0	13.0	3.8	
4	Latrobe River	4.3	11.4	1.8	
19	W. Lake Well.				7.5
22	E. Lake Well.				12.5

Isotherm Analysis on Suspended Sediment Samples

The adsorptive properties of the suspended sediments in the water samples were measured in the same way as for the benthic samples. However, the extended incubation for the August samples was 24 h rather than 48 h used on all the other samples. For the experimental determination of adsorption, the amount of adsorbing material in the water samples (the

suspended sediments) was much less than for the benthic sediment samples. Consequently, for the suspended sediment samples, the amounts adsorbed and the changes in the concentration in the dissolved phase because of adsorption were small and the determination of reliable isotherms was sometimes not possible.

The adsorption measurements for almost all the samples were more scattered and less regular than those obtained for the benthic samples. So, instead of determining a best-fit exponent for Eq. 1 as was done for the benthic samples, it was assumed that the same exponent obtained for the benthic samples sites applied to the suspended sediment samples. In other words, only the second stage in the isotherm fitting was applied to the suspended sediment samples for both the short and long-term incubations. Samples from site 19 collected on January 4, 2001 were analysed at a pH of both 7.1 and 8. The analysis at the higher pH did not produce reliable isotherms. Further reference to the sample from this site pertains to the analysis at pH 7.1. This pH is similar to that measured in Lake Wellington at the time (pH 7.0-7.2). There was negligible sorption for the Site 19 water at pH 8 - even at ambient P concentrations (i.e. no added P), > 99% of the P was in the dissolved phase. This made it impossible to define the isotherm however it does dramatically show the impact of pH.

Figure 4 shows the results of the adsorption experiments and the fitted Freundlich isotherms for the eight samples for which further analysis was undertaken. The label in the top left hand corner of each graph frame refer to the month in which the sample was collected followed by the site number. Note that in Fig. 4, only the isotherms for June are plotted with the same P_s scale as the benthic isotherms shown in Figs. 2 and 3. The January 19 isotherms are plotted with double the scale. All the other isotherms are plotted with much larger scales. Unfortunately, it was not possible to determine reliable isotherms for the samples from the Latrobe River.

Table 5 presents the results of the values of A from fitting procedure on the longer- term incubations (24-h or 48-h). Of these samples, only June 03, June 04, and January 19 have approximately similar values of A to those measured for the benthic sediments presented in Table 3 (up to $A = 109$). The three sediment samples from Lake Wellington (B, C7, and D) have an average A of 52, whereas samples June 03, June 04, and January 19 have an average A of 107. However, the average concentration of Easily Extractable Fe in these three suspended sediment samples is 24 mg g^{-1} versus 13 mg g^{-1} for the benthic sediments. It seems that the higher adsorptive capacity of the three suspended sediment samples can be explained as being due to a proportionally higher concentration of Fe.

Freundlich Isotherms - Suspended Sediments

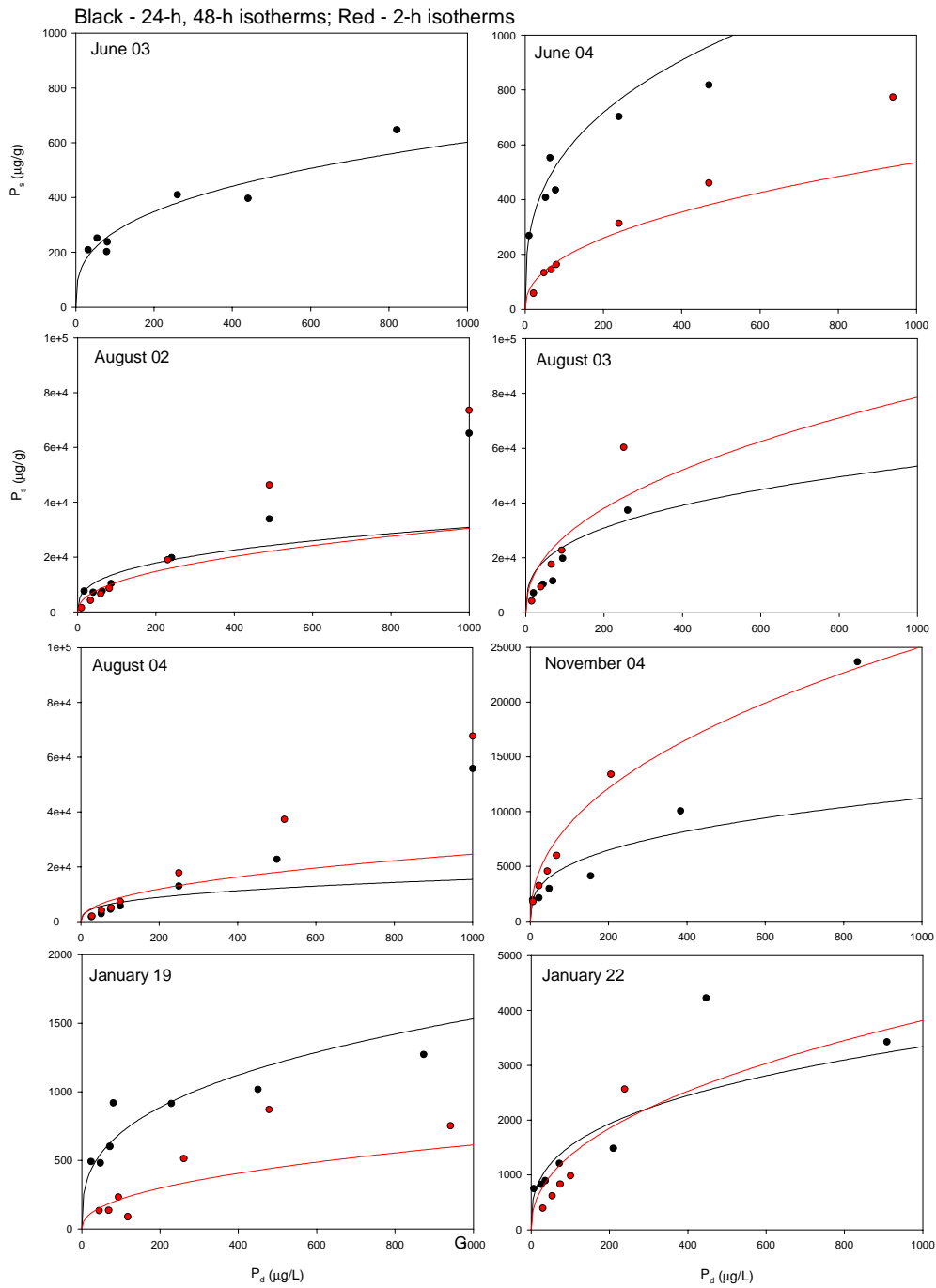


Figure 4. Measurements and fitted Freundlich isotherms for 2-h and 48-h incubations on suspended sediments.

Table 5. Values of the fitted constant *A* for the Freundlich 48-h isotherms determined for the suspended sediments, the concentrations of Easily Extractable Iron, and the ratio of *A* to the concentration of Easily Extractable Iron.

Sample	<i>A</i> (24-h, 48-h)	E. Extr. Fe mg g ⁻¹	Ratio A : E.E. Fe
June 03	58	20	3
June 04	118	31	4
August 02	2947	23	130
August 03	5101	41	125
August 04	1473	39	38
November 04	1071	14	75
January 19	146	20	7
January 22	319	14	23

Also, only the samples from June 04 and from January 19 show a similar relationship between the 2-h and 48-h incubations to that of the benthic samples. For these two samples the 48-h isotherm is of the order of double the level of the 2-h isotherm. We do not have a 2-h isotherm for June 03. It seems that samples June 03, June 04, and January 19, exhibit adsorption behaviour which is similar to that of the benthic sediments in Lake Wellington if account is taken of these suspended sediment samples having approximately double the Fe content of the benthic samples from the Lake.

The other suspended sediment samples shown in Table 5 all have much higher adsorption coefficients, *A*, than do samples June 03, June 04, January 19 and all the benthic samples. The samples collected during the August and November field trips show values of *A* over 1000. None of these samples show particularly high Fe concentrations, which would allow an explanation of their very large adsorptive capacities. For samples August 03, August 04, and November 04, the fitted isotherms for the 2-h incubations were higher than those for the longer incubation in each case. This is behaviour that is very different from what was measured in the benthic samples. It is speculated that the very high sorption found for a lot of the Lake Wellington water samples may be due to the particulate matter being algae as well as abiotic matter - the only discriminant was the 0.2 micron filter. The much lower Fe:dry weight ratio and much higher TP:dry weight ratio for these 'aberrant' samples supports this contention of a relatively high proportion of algal material in the sample. Such algae would readily take up added P.

Conclusions

P adsorption to sediments is a process that occurs over a continuum of time scales. Ideally, for applications to the model it would be desirable to have isotherms determined for timescales that match the timescales of variation in P concentration in the model. These timescales are likely to be weeks or longer so that of the two isotherms we have measurements for, the 48-h isotherms are the better choice. These isotherms are unlikely to represent adsorptive equilibrium, but unfortunately we can't yet estimate to what extent the 48-h isotherm deviates from one determined after a month-long incubation, say. Froelich (1988) has suggested that 50-90% of desorbable P is available in the 'fast-reaction' component which adsorbs/desorbs on the timescale of minutes to hours. His 'slow-reaction' component takes days, months or perhaps years to adsorb/desorb. On this basis, the 48-h isotherm should account for all the fast-reaction component and some of the slow-reaction component as well. It could be expected that the 48-h isotherms would be 50% in error (for representing long-term equilibrium) at worst, but would probably have an error much less than this.

It is suggested that the Freundlich isotherm be used in the model application for both suspended and benthic sediments. The same exponent for all isotherms should be used ($B = 0.34$). Although the variation in the value of the adsorption coefficient (A) fitted to the adsorption measurements on benthic sediments is about a factor of 3, it is suggested that for the sake of simplicity, that $A = 74$ for the benthic sediments in the model. This is the average of the measured coefficients. None of the measured benthic coefficients differ from this value by more than a factor of two. The suspended sediments in Lake Wellington have significantly more iron in them than the bottom sediments and this appears to cause higher adsorption coefficients. The average value of A for the three suspended sediment samples in Lake Wellington having isotherms consistent with adsorption processes is $A = 107$. It is suggested that this value for A be used both for suspended sediments within the Gippsland Lakes and for the suspended sediments in the inflowing rivers.

References

- Froelich, P.N. 1988. Kinetic control of dissolved phosphorus in natural rivers: a primer on the phosphate buffer mechanism. *Limnol. Oceanogr.* **33**, 649-668.
- Golterman, H.L. 1995. Theoretical aspects of the adsorption of ortho-phosphate onto iron-hydroxide. *Hydrobiologia*, **315**, 59-68.
- House, W.A., Denison, F.H., and P.D. Armitage. 1995. Comparison of the uptake of inorganic phosphorus to suspended and stream bed-sediment. *Wat. Res.* **29(3)**, 767-779.
- Longmore, A.R. 2000. Gippsland Lakes Sediment Nutrient Survey. Marine and Freshwater Resources Institute Report No. 19. Fisheries Victoria, Victoria Australia..
- Portielje, R. and L. Lijklema. 1993. Sorption of phosphate as a result of enhanced external loading. *Hydrobiologia* **253**, 249-261.
- Slomp, C.P., Malschaert, J.F.P., and W. Van Raaphorst. 1998. The role of adsorption in sediment-water exchange of phosphate in North Sea continental margin sediments. *Limnol Oceanogr.* **43(5)**, 832-846.

Appendix - Short Progress Report on P Sorption & Particle Sizing

(slightly updated – March 2001)

Prepared by Drs Mike Grace & Soheyl Tadjiki, Water Studies Centre, Monash University

Sampling

At the time of this report (5/9/00), 2 sets of samples have been collected from the Gippsland Lakes for laboratory-based analysis of phosphorus sorption/desorption and particle sizing. Each set of samples is comprised of 4 sites - one just upstream from the mouth of the Latrobe River and 3 at approximately equal distances across Lake Wellington. Some general characteristics of these sites are given in Table 1.

Table 1. Water Quality Characteristics of the 4 Sites in the Gippsland Lakes Study, 2000

Site	Location	June 5		August 2		
		FRP ($\mu\text{g P/L}$)	Salinity (‰)	FRP ($\mu\text{g P/L}$)	Salinity (‰)	Turbidity (NTU)
4 Latrobe River	S 38° 07.124' E 147° 14.419'	10	4.3	7	11.4	8.1
3 Lake Wellington	S 38° 06.771' E 147° 15.916'	10	15.0	1	13.0	4.1
2 Lake Wellington	S 38° 06.064' E 147° 18.669'	9	15.2	2	13.4	2.0
1 Lake Wellington	S 38° 05.341' E 147° 23.929'	19	15.5	1	14.2	0.8

It is interesting to note the high variability in Lake Wellington FRP concentration. The weather during the June sampling was extremely rough (2 m waves) hence resuspension was highly probable. The August sampling conditions were relatively calm.

P Sorption

P sorption isotherms have been completed for all 4 sites for each trip. However, we are waiting upon dry weight measurements - should be available within the next week - which preclude description of sorption on a particle dry weight basis. P partitioning measurements were performed after 2 hours incubation with the added phosphate and again after 24 or 48 hours. The dual time range allows determination of both rapid sorption processes and slower uptake.

The results clearly demonstrate a marked difference between riverine and lacustrine samples. It appears that there is considerably more uptake onto the riverine samples but this may simply be a function of much higher suspended solids load (e.g. see the turbidity results for August). For the lake samples, almost all the added P remained in solution (i.e. in the < 0.2 μm filtrate) during both trips.

Results on particle characterization (total & extractable iron, total phosphorus and total organic carbon concentrations) will be available within the next few days and may help to explain sorption behaviour if the dry weight-corrected data show significant variability.

During January 2001, the Total Fe content in the solid material (TSS) was typically approx. 5% by weight. During August 2000, the Fe content in the 3 Lake Wellington samples (1-3) was dramatically less, providing further evidence to the contention that most of the TSS was in fact biotic material (algae). Having a high biotic content may explain the very high sorption maximum for these (august) samples. The Total P content in the 3 August samples was much higher (0.50-0.55% by dry weight) than found for all the other samples (0.07-0.27%).

Particle Sizing

Particle sizing measurements have been completed on all 8 samples as well. The video capture technique is limited to quantitatively observing particles greater than 4 μm in size. Particle size distributions were measured at 3 times after the samples were initially shaken vigorously to ensure complete mixing and resuspension of material that might have settled during transport and storage: immediately after shaking, 1 hour after shaking and again after 24 hours. Between measurements, samples were stirred at a rate just sufficient to prevent settling.

The experiments on the June samples have demonstrated that the particle size distribution behaves differently at each site. The zero time distributions of all samples were effectively the same. The riverine and the 2 lake samples furthest from the river showed little variation in distribution with time. However, the lake sample closest to the river showed a broadening of the distribution and an increase in the peak particle size to ca. 8 μm diameter. There is also a suggestion of a broadening distribution after 24 hours in the mid-lake sample.

Work is continuing on the interpretation of these distributions.

Future Work

Two further sample sets will be collected for us (by the Victorian EPA during their regular trips to the Lakes) for P sorption and particle sizing studies. Sediment samples from Lakes King, Victoria and Wellington will also be analysed

Examination of Water Balance Scenarios for Locks Across Lakes Entrance and McLennans Strait

Ian T. Webster
CSIRO Land & Water

September 2001

It has been suggested that the environmental condition of the Gippsland Lakes would be improved if they could be returned to freshwater systems. This objective could be accomplished by constructing locks across the two narrowest sections of the Lakes namely McLennans Strait and Lakes Entrance. A lock across McLennans Strait that allowed passage of freshwater flows but not return saline flows from Lake Victoria would turn Lake Wellington fresh. Similarly, a lock constructed across Lakes Entrance that prevented the ingress of seawater from Bass Strait would cause the entire Lakes' system to turn fresh.

Such locks that prevent seawater from entering the Lakes have implications for water levels within the Lakes. The water level within a freshwater lake is maintained by a balance between the river discharge into the lake, evaporation from the lake's surface, precipitation, and the outflow to a neighbouring lake or the sea. If the net amount of water entering the lake is positive then the water level will rise, and conversely negative net input will cause the water level to drop. Here, we examine the implications for water levels within the Lakes for locks constructed across McLennans Strait and across Lakes Entrance.

Water Balance Equation

The water balance within a lake can be expressed as:

$$A \frac{dH}{dt} = Q_{in} - AE + AP - Q_{out}$$

In this equation, A is the area of the lake, H is the water level, t is time, Q_{in} is freshwater inflow, E is the evaporation rate, P is precipitation rate, and Q_{out} is the outflow. The left hand side of this equation represents the rate of change in the volume of the lake associated with the water level change and the right hand side equals the net amount of water entering the lake. This equation can be integrated forward in time to obtain water levels using prescribed freshwater discharges, precipitation and rainfall. When the lake is full, we assume that $H = 0$ as a reference height. This is assumed to be its maximum level; the outflow is sufficient to maintain the height at this level. Further if $H < 0$, then we assume that the outflow is zero; that is, the lake is allowed to fill. With these conditions the specification of the outflow is not required.

McLennans Strait Lock

For the case of a lock across McLennans Strait, the lake considered is Lake Wellington, which has an area of 148 km². The inflow, Q_{in} is the sum of the measured discharges from the Latrobe, Thomson, and Avon Rivers. The evaporation and precipitation rates used in the

analysis are those measured at East Sale Airport. The analysis extends for 23 years between 13 November, 1976 and 12 September, 1999. Figure 1 shows the predicted water levels in Lake Wellington using measured river discharges. As part of the Gippsland Water Quality Modelling Project a number of water harvesting scenarios are being considered. One of these scenarios has the discharges of all the rivers flowing into the Gippsland Lakes reduce by 20%.

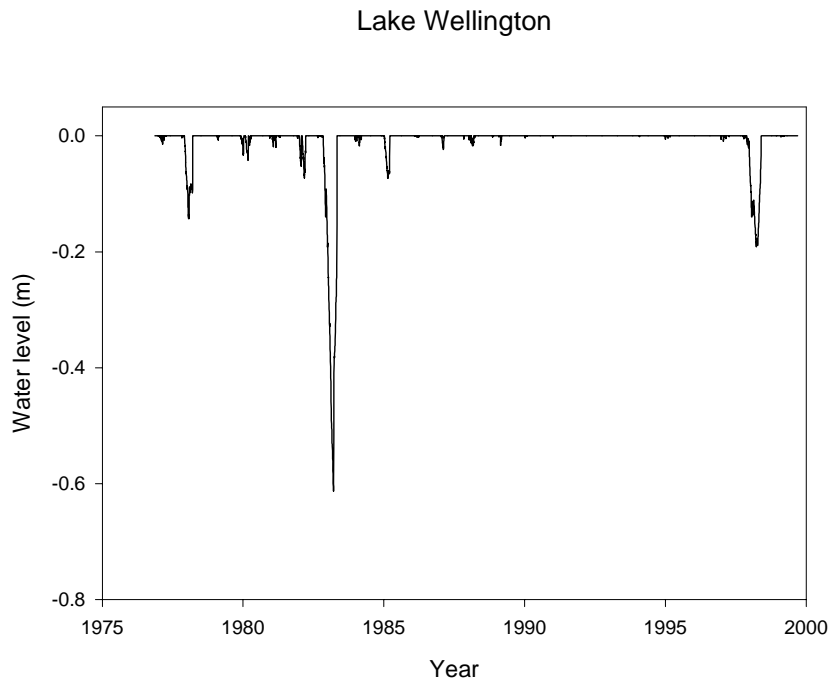


Figure 1. Predicted water levels in Lake Wellington for the case of a lock constructed across McLennans Strait.

In most summers, there would be at least a short period of time when input into Lake Wellington due to river discharge and precipitation are not enough to replace the water lost by evaporation. This shortfall causes the water level to drop below its 'full' level, but the drop would be almost always less than 0.1m. The water level would drop by more than 0.1m on only three occasions namely during the summers of 1977/78, 1982/83, and 1997/98. During the major drought of 1982/83 the drop in water level of 0.61m would be much larger than in the other years. The water harvesting scenario involving a 20% increase in abstraction results in the predicted water level drop increasing to 0.67m.

Lakes Entrance Lock

For the case of a lock being constructed across Lakes Entrance, the body of water considered in the analysis is Lake Wellington, Lake Victoria, Lake King, Reeve Channel and Jones Bay taken together. The combined area of these Lakes is 361 km². Lake Reeve is excluded since it only has water in it during times of elevated river discharge when the Lakes are all full anyway. The rivers discharging into the system include those flowing into Lake Wellington as well as the Mitchell, Nicholson, and Tambo Rivers. Evaporation and precipitation rates are obtained from East Sale Airport as for the Lake Wellington analysis. Figure 2 shows the predicted water levels for the Lakes Entrance case.

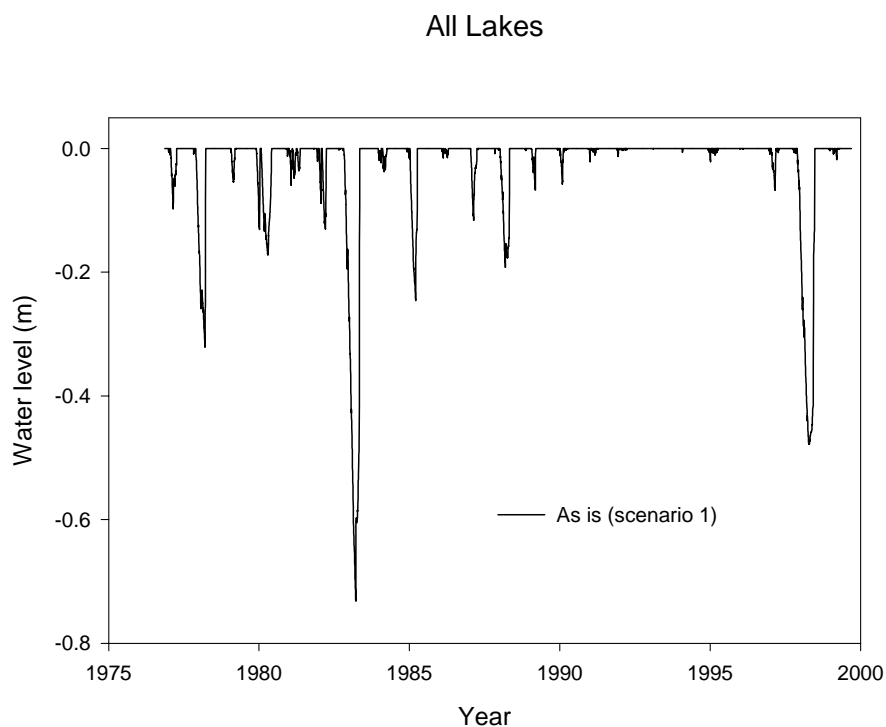


Figure 2. Predicted water levels in the Gippsland Lakes for the case of a lock constructed across Lakes Entrance.

The lock at Lakes Entrance is predicted to cause more significant water level changes within the Lakes than the changes in Lake Wellington that would be caused by a lock across McLennans Strait. Although the level in the Lakes would drop by less than 0.10m in most years, during the summers of 1977/78, 1982/83, and 1997/98, the drop would exceed 0.3m. As for the McLennans Strait lock, the largest drop for the lock across Lakes Entrance (0.73m) is predicted to occur during the summer of 1982/83.

Water level changes occur throughout the Lakes in response to low-frequency sea level changes, winds, tides, and to variation in river discharge (Walker and Andrewartha, 2000; Grayson et al., 1999). The range in water level caused by these phenomena is about 0.3m so it is suggested that water level drops caused by restricting water exchange by locks should probably not be considered significant unless they are larger than 0.3m. By this criterion, the construction of a lock across McLennans Strait would cause significant water level changes in Lake Wellington in one year out of 23 (approximately 4% annual exceedence probability). The Lakes Entrance lock would cause significant water level drops in three years out of the 23 (approximately 13% annual exceedence probability).

The largest part of the natural variability in water levels within the Lakes is caused by low-frequency water level variations in Bass Strait. With the construction of locks these fluctuations would no longer contribute to water level variability within all or part of the Lakes' system. In effect, lock construction would lead to smaller water level variation during most years, but during times of drought there would be an opportunity for variation that is larger than the natural condition.

Discussion

The weakest part of the analysis is probably the reliance on the evaporation measurements from East Sale Airport. These measurements were made from an evaporation pan that may not yield accurate estimates of evaporation rates from the large open water surfaces of the Lakes. However, as part of their hydrodynamic modelling of the Lakes, Walker and Andrewartha (2000) used the measured evaporation rates in order to predict water temperatures. In the first year of the simulation, measured temperatures were well predicted by the model, but in the second year predicted temperatures were too high. These comparisons suggest that measured evaporation rates were consistent with the overall heat balance of the Lakes in the first year, but in the second year measured evaporation rates were too low.

Inevitably, the construction of locks to prevent exchange with the sea would have huge impacts on the ecology of the Lakes. An obvious major impact is the replacement of the biota adapted to variable salinity by a freshwater ecology. The construction of locks would certainly reduce the flushing rate of the Lakes. However, it could be that the elimination of saline inflows through the Entrance would cause the Lakes to have the tendency to stratify in temperature only. Persistent stratification would not occur during Autumn and Winter and for the rest of the year it is possible that stratification would only occur for periods of limited duration. With the maintenance of the generally higher oxygen concentrations in bottom water that would accompany more sporadic stratification, we might expect denitrification efficiencies to be higher in Lakes King and Victoria. The reduction in flushing efficiency caused by lock construction could be more than compensated by an increased loss rate for nitrogen through denitrification. It is not known to what extent slow seepage from the ocean (or salty water from the Macalister Irrigation District groundwater area) would enable the formation of saline stratification and anoxic bottom waters. The prevention of saline exchange with the ocean is likely to impact on turbidity particularly in Lakes King and Victoria. If these two Lakes were fresh water then flocculation would be less effective at reducing their turbidity than is the present case.

The analysis has not considered the detailed effects the lock structures on the flows past them. Inevitably, the structures will impose some restriction of these flows. Any such constriction will exacerbate the problem of flooding during times of high discharge of the rivers flowing into the Lakes.

References

Grayson, R.B., McMaster, M., McCowan, A., 1999. Summary of work on the Gippsland Lakes by the Centre for Environmental Applied Hydrology, 1998/99. CEAH report 1/99, University Of Melbourne.

Walker, S. and Andrewartha, J. 2000. Gippsland Lakes Environmental Study – Hydrodynamic Modelling. CSIRO.

First-cut Estimation for Seepage across Bunga Arm Barrier

Kim Seong Tan
Centre for Environmental Applied Hydrology
Department of Civil & Environmental Engineering
The University of Melbourne
Victoria 3010, Australia

Supervisors: A/Prof. Rodger Grayson & Dr. Andrew Western

July 2000

Introduction

The Gippsland Lakes is a typical coastal lagoon located in the south-eastern part of Australia. One of the unique characteristics of a coastal lagoon is that the lagoon basin, usually shore-parallel, is separated by a narrow strip of coastal barrier/dune. In the case of the Gippsland Lakes, the barrier separating the Bunga Arm from the coastal ocean is sandy in nature and appears to be quite “narrow” and “long”.

As part of the Gippsland Lakes Environmental Study modelling work, it is necessary to carry out an overall appraisal of the environmental system, so that “important” features of the system are not missed out in the model representation. As far as the possibility of seepage across the Bunga Arm barrier is concerned, we reckoned that at least a simple preliminary investigation is warranted. Consequently, the following first cut estimation has been carried out to check if seepage is “important” enough to be included in the system dynamics of the model set-up.

Physical Setting

The Bunga Arm barrier is a “narrow” and “elongated” sand dune formed by coastal processes subsequent to the Holocene sea level rise, with widths ranging from 100 to 300m and a length of about 15km. As part of the system dynamics of the Gippsland Lakes, the Bunga Arm is a shallow (~1 to 2m), narrow (~100 to 400m) and elongated (~15km) water body which is hydrodynamically well connected to the main water body of the larger Lake Victoria/King. All dimensions are approximated from the topographical map “*1:100,000 Series Sheet 8422-Bairnsdale, Victoria Edi. 1*” published in 1976 by NATMAP, Commonwealth of Australia.

Soil Profile

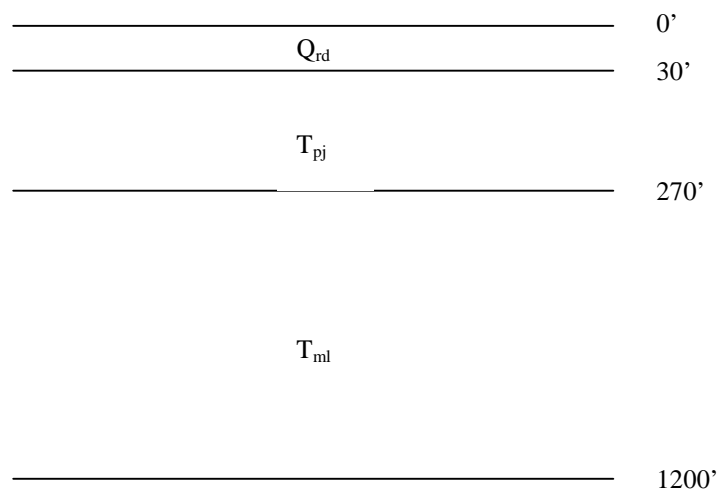
Based on the “*NMP/60/122 Atlas of Australian Soils: Melbourne-Tasmania Area-Sheet 2*” published in 1961 by the Div. of Natational Mapping, Dept. of Development Canberra, the dominant soil in Bunga Arm is of “Type B3-Siliceous sand with uniform texture profile”.

The “*Australia 1:250,000 Geological Series Sheet SJ55-11 Sale 1st Edi.*” published in 1972 by the Geological Survey of Victoria shows a soil profile with a transect across Sperm Whale Head, which is the soil profile nearest to Bunga Arm barrier available to us. We assume that this profile is representative to the Bunga Arm barrier.

0' to 30'	Q_{rd} : Dunes, beach deposits, Aeolian sand sheets, raised beaches, estuarine, sand, barrier deposits and ridges, shell beds.
30' to 270'	Jemmy's Point Formation - T_{pj} : Calcareous and siliceous sand, sand sheets and ridges, dune limestone, clay, gravel.
270' to 1200'	Gippsland Limestone - T_{ml} :Limestone and marl, ferruginous sand.

Based on these information, we schematized the soil profile for the purpose of this preliminary investigation as follows (Figure 1):-

Figure 1 : Schematised Soil Profile (N.T.S.)



Based on the soil profile above, we consider that the first 30' (~10m) of sand dunes/beach deposits as being “conductive” for the purpose of this preliminary seepage investigation.

Coefficient of Permeability

For clean sands and sand gravel-sand mixtures, the coefficient of permeability, k is around 10^{-1} to 10^{-4} m/s; whereas for very fine sands, silts and clay-silt laminate, k is around 10^{-4} to 10^{-7} m/s (Table 2.1 in Craig, 1997). We would use an average value of $k = 10^{-4}$ m/s.

Another method of estimating conductivity for sandy soil, as recommended by Hazen, is $k = 10^{-2} D_{10}^2$, where k is in m/s and D_{10} in mm. Assuming $D_{10} = 0.1$ mm for the sand of the Bunga Arm barrier yields $k = 10^{-4}$ m/s.

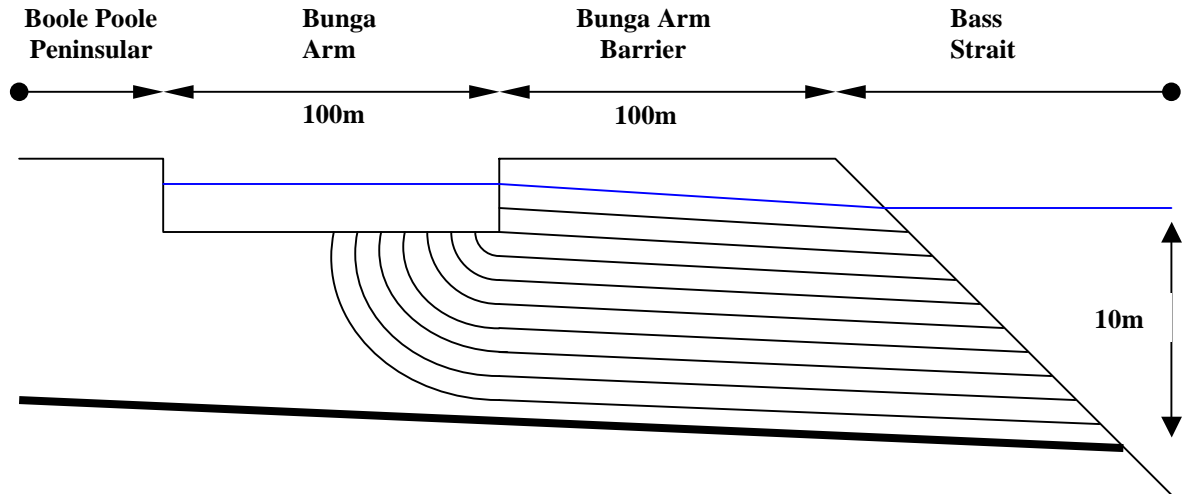
Difference in Total Head

The maximum difference in total head between the ocean and the Lakes is taken as 0.5m.

Seepage through the Bunga Arm Barrier

For a preliminary seepage investigation, the Bunga Arm barrier could be schematically represented in Figure 2.

Figure 2 : Schematised Bunga Arm Barrier with Flow Net (N.T.S.)



Based on the simple flow net method with curvilinear squares (Craig, 1997), seepage per unit width,

$$q = (N_f/N_e) k h = (10/100) 10^{-4} h = (10^{-5} h) \text{ m}^3/\text{s per m width}$$

where

- N_f = number of flow channel = 10
- N_e = number of equipotential drops along each of the channel = 100
- k = coefficient of permeability, assuming isotropic condition = 10⁻⁴ m/s
- h = difference in total head = 0.5m

For a dune of length, L = 15km, the seepage rate across the entire dune is thus,

$$Q_{BA} = q \times L = (10^{-5} h) \times 15,000 = (0.15 h) \text{ m}^3/\text{s}$$

Table 1: Seepage through the Bunga Arm Barrier

Head Difference, h (m)	Discharge, Q _{BA} (m ³ /s)	Time Weighted Average Discharge, Q _{BA(ave)} (m ³ /s)
0.1	0.015	0.0150
0.2	0.030	0.0225
0.3	0.045	0.0300
0.4	0.060	0.0375
0.5	0.075	0.0450

For h = 0.5m, the time weighted average discharge, Q_{BA(ave)} = 0.045m³/s.

The same analysis can be performed with more extreme values for k (10^{-3} m/s), head of 2 m, length of dune over which seepage occurs (say 50km to include lake Reeve), narrower dune and taking just the peak flow rate. This extreme case analysis results in flow rates of $10 \text{ m}^3/\text{s}$. As will be seen below, this is still very small compared to flow rates at the entrance.

Discharge through Lakes Entrance Waterway

The Lakes Entrance has the following characteristics :-

- Depth, $D = 4.5\text{m}$
- Breadth, $B = 80$ to 120 m (use 80m)
- Length, $L = 400\text{m}$
- Mannings Roughness Coeff, $n = 0.03$

Thus, discharge through Lakes Entrance waterway,

$$Q_{LE} = (1/n) A R^{2/3} S^{1/2} = (1/0.03)(4.5 \times 80)(4.5 \times 80/89)^{2/3} (S)^{1/2} = 30464 S^{1/2} \text{ m}^3/\text{s}$$

Assuming the entire Lakes with a total surface area, $A_{Lakes} = 350\text{km}^2$ is responding with the low frequency ocean level fluctuation in a pumping mode, the total time required for the entire Lakes water level to raise/drop by 0.5m is approximately 65.3hr .

Table 2 : Discharge through the Lakes Entrance Waterway

Head Difference, h (m)	Water Surface Slope, S	Discharge, Q_{LE} (m^3/s)	Velocity, V_{LE} (m/s)	Time to Discharge (hr)	Cumulative Time (hr)
0.1	0.00025	482	1.3	20.2	20.2
0.2	0.00050	681	1.9	14.3	34.5
0.3	0.00075	834	2.3	11.7	46.2
0.4	0.00100	963	2.7	10.1	56.3
0.5	0.00125	1077	3.0	9.0	65.3

The approximated maximum magnitude of the low frequency ocean level fluctuation is in the order of 0.5m ($\sim 50\text{mbar}$), lasting for between 1 to 10 days. A change of $h = 0.5\text{m}$ in the Lakes-ocean water level that takes $T_{0.5\text{m}} = 65.3\text{hr}$ ($\sim 2.7\text{days}$), enabling a complete volume exchange to occur would be a common phenomenon.

Significance of Seepage

We chose to evaluate the significance of seepage by simply computing the percentage ratio of seepage through the Bunga Arm barrier and discharge through the Lakes Entrance waterway.

Under the assumed scenario, the volume of seepage through Bunga Arm barrier,

$$V_{BA} = Q_{BA} \times T_{0.5\text{m}} = 0.045\text{m}^3/\text{s} \times 65.3\text{hr} \times 3600\text{s/hr} \sim 10,580\text{m}^3$$

Whereas the volume of discharge through the Lakes Entrance waterway,

$$V_{LE} = A_{Lakes} \times h = 3.5 \times 10^8 \text{m}^2 \times 0.5 \text{m} = 1.75 \times 10^8 \text{m}^3$$

$$V_{BA} / V_{LE} = (10,580 \text{m}^3 / 1.75 \times 10^8 \text{m}^3) \times 100\% = 0.006\%$$

Even with the extreme assumptions, this ratio is only a few percent.

Conclusion

In this preliminary analysis, we generally assume the average to maximum values for the relevant parameters involved. For example the water level difference of 0.5m (as a result of low frequency forcings) is considered as a reasonable assumption; the coefficient of permeability for sand may be an average value; whereas the depth of the permeable soil could have been several times thicker than the 10m assumed.

Even if the assumed values are out by, several orders of magnitude, the resultant percentage of seepage through the Bunga Arm barrier is still less than a few percent, which is a relatively “insignificant” proportion for the Lakes system as far as the hydrodynamic and water quality modeling is concerned.

Based on the above simplified seepage analysis and the information available to us, we therefore concluded that the seepage through the Bunga Arm is not “important” enough to be included in the system dynamics of the hydrodynamic and water quality modeling set-up.

Pre-European load estimates into the Gippsland Lakes

Rodger Grayson, KS Tan, and Stephen Wealands

Centre for Environmental Applied Hydrology, Department of Civil and Environmental Engineering, University of Melbourne

September 2001

Overview

At the Steering Committee meeting and Gippsland Lakes Modelling Workshop of Wednesday 22nd August, it was decided that estimates should be made of pre-European loads of sediment, total nitrogen and total phosphorous into the Gippsland Lakes. It was recognised that such estimates are subject to a large degree of uncertainty, yet it was determined that they may be useful for putting possible management scenarios into perspective. The Centre for Environmental Applied Hydrology undertook to assemble estimates based on easily available information.

As noted at the meeting of August 22, there have been a large number of other changes in the Lakes ecosystem since European settlement, including the permanent opening of the entrance. The impact of these in conjunction with changed loads would be needed to establish the pre-European ecological behaviour of the system, and even then, one may question their relevance to setting present-day management objectives.

Methods

Two approaches were used to make these estimates.

Method 1 was to simply assume that the overall catchment area of the Gippsland Lakes were once forested, and then apply published average loading rates (in Tonnes/km²/yr) to the areas of the Eastern and Western Rivers (draining into Lakes King and Wellington respectively). There is a large range of published generation rates for forests, and the lower end of this range was used.

Method 2 was to:

1. Get the current percentage of major land uses in each of the six major river catchments (split into urban, forested uplands, forested lowlands, irrigation, dryland grazing, horticulture). The map used was the NRE 1:250,000 coverage from their data library (Figure 1).
2. Use our present average annual computed loads to solve for the average annual generation rates from these land uses. This required the application of an optimisation procedure where each generation rate was constrained by the published range of values. This was done on each catchment individually but the results grouped as “Eastern” and “Western” rivers. This approach allowed us to have more confidence in the load generation factors used to estimate pre-European conditions.
3. Assume that the pre-European situation was forested lowlands and uplands and compute long term average loads.

The results were then compared to other estimates from Grayson et al. (1994) for the Latrobe River catchment and some preliminary results available from the National Land and Water Resources Audit (NLWRA). These latter results are as yet unpublished and were provided for our interest (Warwick McDonald, Bill Young pers. Comm.).

Results

Table 1. Estimates using Method 1 - direct estimates of loadings from published data, and assuming 100% forest cover pre-European

	Present total loads (this study)	Present average loading	Assumed pre-European loading	Ratio of present to pre-European loading	Approx. reduction in present loading needed to reach pre-European rates
Western Rivers (area ~ 11000 km²)	(T/yr)	(T/km²/yr)	(T/km²/yr)		%
TSS	170000	15.6	5	3.1	70
TN	2000	0.18	0.08	2.2	55
TP	230	0.02	0.005	4.0	75
Eastern Rivers (area ~ 9000 km²)					
TSS	49000	5.4	5	1.1	10
TN	780	0.09	0.08	1.1	10
TP	75	0.008	0.005	1.6	38
Overall (area ~ 20000 km²)					
TSS	220000	11	5	2.2	55
TN	2800	0.14	0.08	1.8	43
TP	300	0.015	0.005	3.0	67

Table 2. Estimates using Method 2 - optimised values of generation rates based on present land use (see text for explanation)

	Present total loads (this study)	Computed pre-European loads	Ratio of present to pre-European loads	Approx. reduction in present loading needed to reach pre-European rates
Western Rivers (area ~ 11000 km²)	(T/yr)	(T/yr)		%
TSS	170000	74000	2.3	57
TN	2000	1100	1.8	45
TP	230	46	5	80
Eastern Rivers (area ~ 9000 km²)				
TSS	49000	46000	1.1	10
TN	780	730	1.1	10
TP	75	60	1.3	23
Overall (area ~ 20000 km²)				
TSS	220000	121000	1.8	44
TN	2800	1900	1.5	33
TP	300	106	2.8	65

Table 3. Comparison of results and recommendations

	Estimated ratios, Method 1	Estimated ratios Method 2	Estimated ratios by others	Proposed ratio for purposes of final reporting	Approx. reduction in present loading needed to reach pre-European rates
Western Rivers (area ~ 11000 km²)					
TSS	3.1	2.3	2 ^a	2.3	55
TN	2.2	1.8	1.1 ^{b,c}	1.8	45
TP	4.0	5	4-5 ^a , 1.3 ^{b,c}	4	75
Eastern Rivers (area ~ 9000 km²)					
TSS	1.1	1.1	-	1.1	10
TN	1.1	1.1	1.1 ^b	1.1	10
TP	1.6	1.3	1.2 ^b	1.3	25
Overall (area ~ 20000 km²)					
TSS	2.2	1.8		2	50
TN	1.8	1.5		1.5	35
TP	3.0	2.8		2.8	65

^a from Grayson, 1994 – “Mitigation of sediment and nutrient movement from sources in the Latrobe River catchment into the Gippsland Lakes”. LWRRDC Project UME8. Centre for Environmental Applied Hydrology. 225p

^b unpublished estimates from the National Land and Water Resources Audit (McDonald, Young, per comm.)

^c this value does not fully account for point sources or intensive agricultural outputs, particularly in the Latrobe River catchment – it would appear that they are underestimated.

Discussion

These estimates are crude and must be accepted as such. There appears to be no better methods for the computation of pre-European loads that are possible without embarking on a major research project, and even then, considerable uncertainty is likely to remain.

The analysis indicates that suspended sediment load delivered to the Lakes is presently around twice the pre-European values, total phosphorous loads have increased by a factor of around three, with total nitrogen being approximately 1.5 times pre-European conditions. It must be noted that these generation rates relate to delivery *to the Lakes* and it is likely that generation rates of sediment back in the catchment are many times greater than pre-European, with greatly increased storage of sediments on flood plains and in-stream storages (there is a significant body of literature discussing this issue). Indeed work done under the NLWRA indicates that *generation* rates of sediment are likely to be from 3 to 100 times pre-European values across the catchments of the Gippsland Lakes.

With respect to phosphorous loads, there will be an increase in loads resulting purely from the increase in sediment, since P is part of soils. But in addition to this, there are elevated P loads resulting from point sources (such as sewage treatment plants, industrial discharges), and increased concentrations in runoff due to fertiliser, increased stocking rates etc. It is therefore expected that TP will have increased more than TSS since European settlement and more in the Western than the Eastern catchments. The composition of the TP load will also have changed and it is likely that bio-available forms of P are relatively more abundant in present day loads. This change in composition of the loads cannot be quantified through this simple analysis, but will have significant ecological impacts.

In the estimates above, it is interesting to note the smaller apparent increase in nitrogen loads following European settlement. Literature values of nitrogen loads from forested areas are not greatly different to those from dryland pasture, nor from the average computed loadings for the catchment based on present data, particularly for the Eastern rivers (see Table 1). It is therefore not surprising that the methods give roughly the same result, since optimised values in Method 2 approach the lower limit of published values. The NLWRA methods are somewhat more sophisticated in relation to TN, yet their values are similar to those of Methods 1 and 2 for the Eastern rivers (where point sources are limited) so we might assume that, to the best of our present knowledge, our estimates are reasonable. The NLWRA estimates for Western Rivers would appear to not fully account for point sources so the value is similar to that of the Eastern Rivers. The recommended value takes better account of point sources.

As with TP, it is likely that there will have been changes in the composition of the nitrogen load, with present loads likely to contain more bio-available forms. The relative difference in changes of TP and TN loads also implies changes in the P:N ratios, which again may have significant ecological impacts. From an ecological point of view, the ratio of interest is the ratio of bio-available forms, which will likely be different to the change in ratio of TP:TN.

Clearly there are many unresolved issues that arise from the analysis presented here, particularly in relation to how the changes in TP and TN relate to changes in bio-available forms. As noted in the introduction, there is also considerable uncertainty in the pre-European estimates themselves. Nevertheless, the results in Table 3 should serve the purpose of the Steering Committee's request to provide some context for proposed management actions.

Figure 1 NRE Land use – land uses were estimated as follows: Urban from 1:100,000 maps; Horticulture from NRE land use *horticulture*; Irrigated land from NRE land use *Pasture-irrigated*; dryland pasture from NRE land use *pasture-dryland*; Forest from total NRE map area minus the other land uses. Lowland and upland forest was distinguished by the 50m contour.

