2.4 GIPPSLAND LAKES, VICTORIA

Ian T. Webster and Phillip W. Ford

Site Area Description

The Gippsland lakes (Site No. 24.; 147.00E, 38.00S) are three interconnected coastal lakes in Victoria, Australia (Figure 2.14; see Figure 1.1). Their area (400 km^2) and large catchment (20 000 km^2) places them amongst the largest Australian coastal lake systems. Their most important physical characteristics are set out in Table 2.12 and their geomorphological and geological setting is detailed in Bird (1978).

Figure 2.12 The Gippsland Lakes region of Victoria.

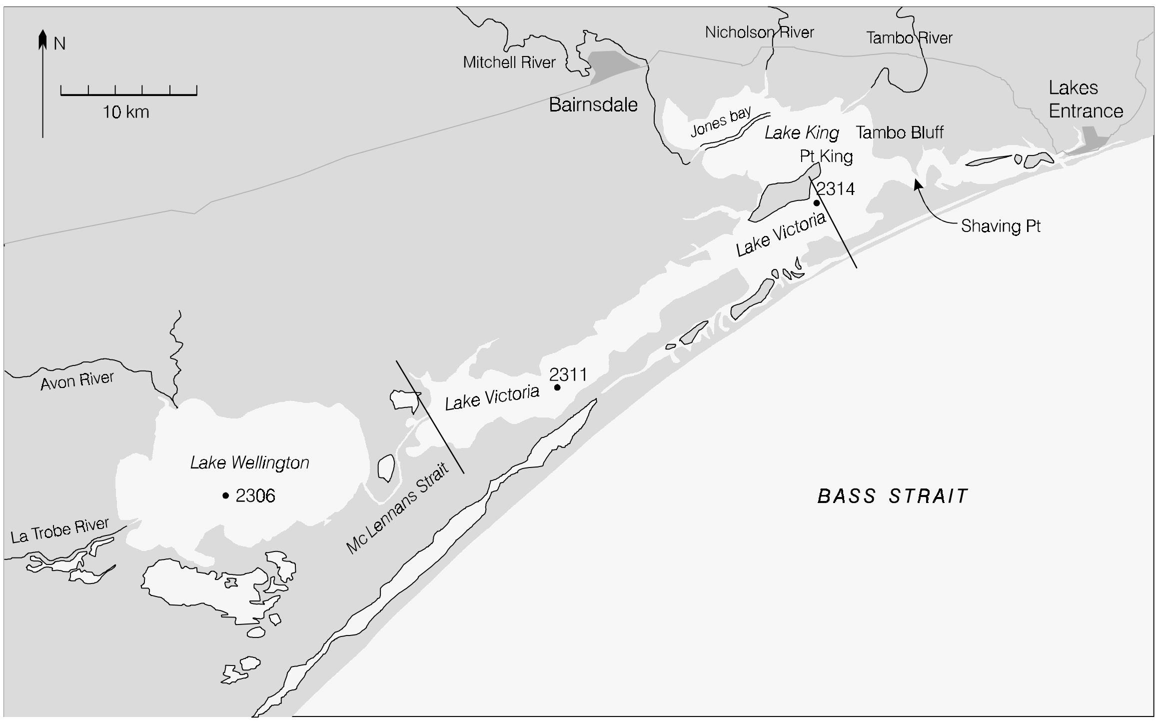


Table 2.12 Physical dimensions of the major Gippsland lakes. (Data from Bek and Bruton 1979.)

Compartment	Mean Length (km)	Mean Width (km)	Surface area (km^2)	Mean depth (m)	Max. Depth (m)	Volume (10^6 m^3)
Lake Wellington	17.8	9.7	148	2.6	6	385
McLennans Strait	9.7	0.2	2	5	8	10
Lake Victoria	27.4	3.2	75	4.8	9	343
Lake King	12.9	6.5	98	5.4	10	524

Originally the lakes were a series of brackish coastal lagoons only intermittently connected to the sea, but in 1889 a permanent entrance to the sea was built at Lakes Entrance. As a consequence, the mean water levels in the lakes were lowered and the water level range decreased; salinity increased in all lakes. The salinity effects are most pronounced in summer as the average annual evaporation (1200 mm) exceeds the precipitation (660 mm) (Robinson 1995). Rainfall and river inflows are greatest in winter.

The lakes are an important recreational resource and also encompass internationally recognised migratory bird sanctuaries, wetlands and national parks. The ecosystem is still adapting to the changes associated with an estuarine environment but is perceived to be degrading with a loss of submerged macrophytes (around the fringes of Lake Wellington), blooms of *Nodularia spumigena* increasing water turbidity, and declining catches in both the commercial and recreational fisheries, all contributing to a public perception of a degraded ecosystem.

The catchment has undergone major changes. Originally heavily forested, the coastal plains and lower slopes have been largely cleared for agriculture including irrigated agriculture and grazing. The flows in the major rivers entering Lake Wellington (Latrobe, Macallister and Thomson rivers) are controlled by storages. The rivers flowing into Lake King are unregulated. Extensive planting and harvesting of native and plantation timber continues in the catchment. A major brown coal-fired power generation industry, and a large paper mill are located on the Latrobe River as are the major towns of the region (total population 130 000). For about 50 years, cooling water, mine water and other effluents plus treated sewage effluent were discharged to the Latrobe River. This has now been diverted from the river. Goldmining in the 19th century has left numerous actively-eroding scars on the landscape and is thought to be the source of high mercury concentrations in the lake sediments.

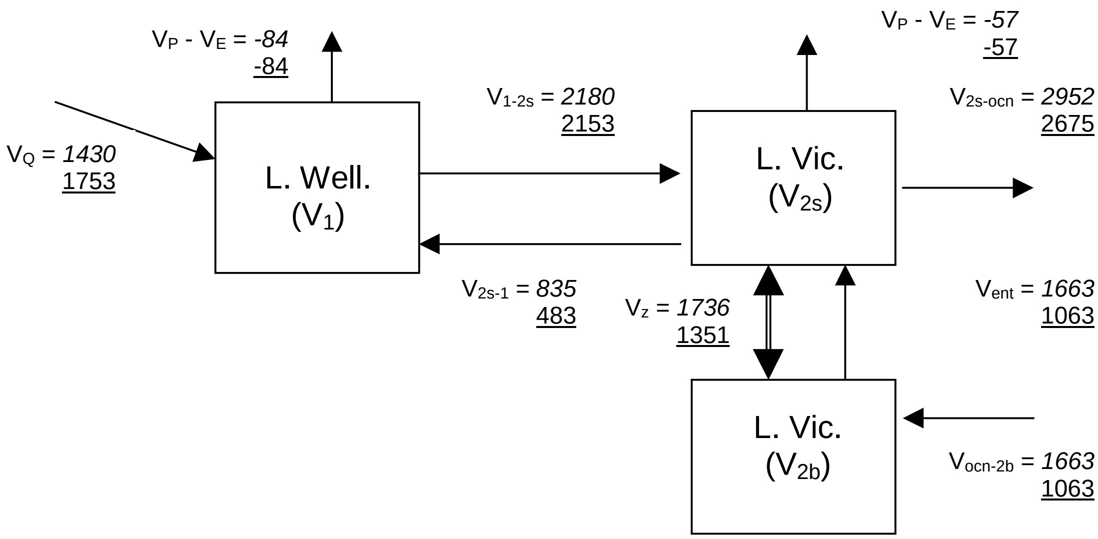
There have been numerous investigations into various aspects of the behaviour of the lakes' ecosystems, often triggered by nuisance algal blooms. A recent review (Harris *et al.* 1998) provides an entry point to this earlier work. There has not been, however, a well-structured long-term monitoring program of nutrient inputs to the lakes, or an integrated investigation of the ecodynamics of the system as a whole. The calculations reported here are based on data supplied by Mr. David Robinson (EPA Victoria) and his help is gratefully acknowledged.

Water and Salt Budgets

Figure 2.15 gives the water and salt budgets for the combined Wellington-Victoria system. The calculations for the years 1988-1990 (July to July) assumed an inflow salinity of zero psu. Runoff is based on weekly measurements of the flow of all the major rivers. Groundwater flow is assumed to be zero. Rainfall is estimated to be 88 x 10^6 m^3 $year^{-1}$ and evaporation 172 x 10^6 m^3 $year^{-1}$. The average water residence times were: 84 days (1988-89) and 92 days (1989-90). The exchange time of Lake Victoria alone was about 45 days in 1988-89 and 49 days in 1989-90. Lake Wellington is shallow and is treated as well-mixed. Lake Victoria, on the other hand, is stratified and there is a two-way salt exchange between Lake Victoria and both Lake Wellington and Lake King.

WATER BUDGET (10^6 m^3/yr)

July 1988 - July 1989
<u>May 1989 - May 1990</u>



SALT BUDGET (10^9 kg/yr)

July 1988 - July 1989
<u>May 1989 - May 1990</u>

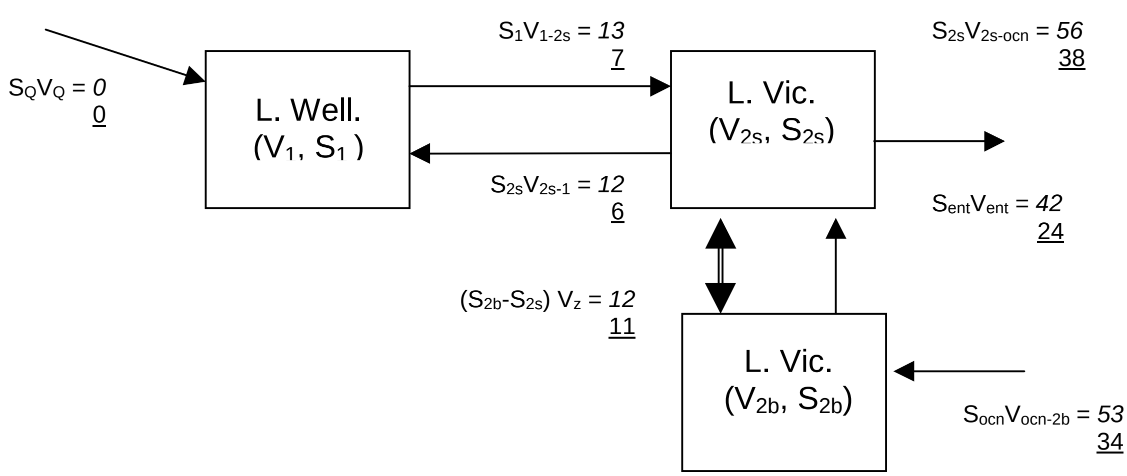


Figure 2.13 Water and salt budgets for July 1988 to May 1990, Lakes Wellington and Victoria.

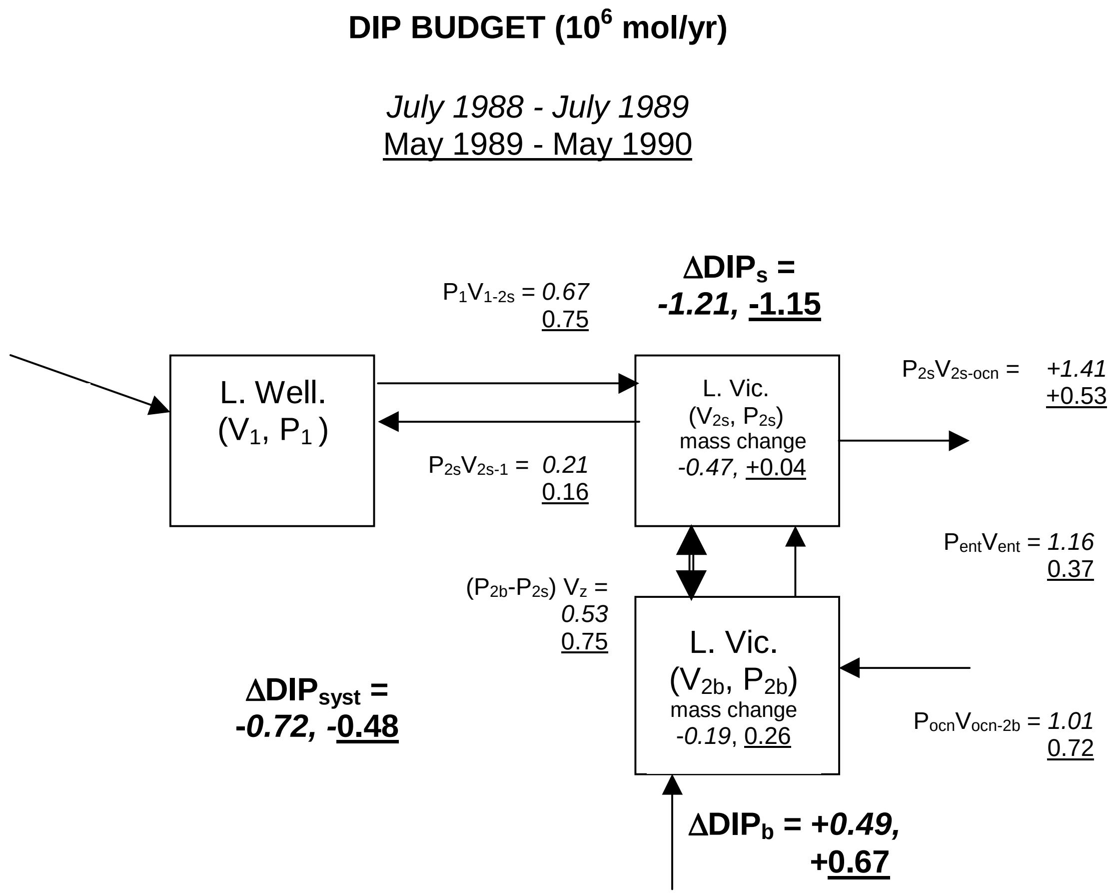


Figure 2.14 DIP budget, Lake Victoria. (DIP concentration of water entering Lake Wellington is not known, although the DIP concentration of water in the lake is known. Therefore the water and salt budgets (Figure 2.13) are used to constrain the exchanging water flow between Lakes Wellington and Victoria. Moreover the changing mass of DIP in Lake Victoria is too large to be treated as 0, so this change is included in the budget.)

DIN BUDGET (10^6 mol/yr)

July 1988 - July 1989
May 1989 - May 1990

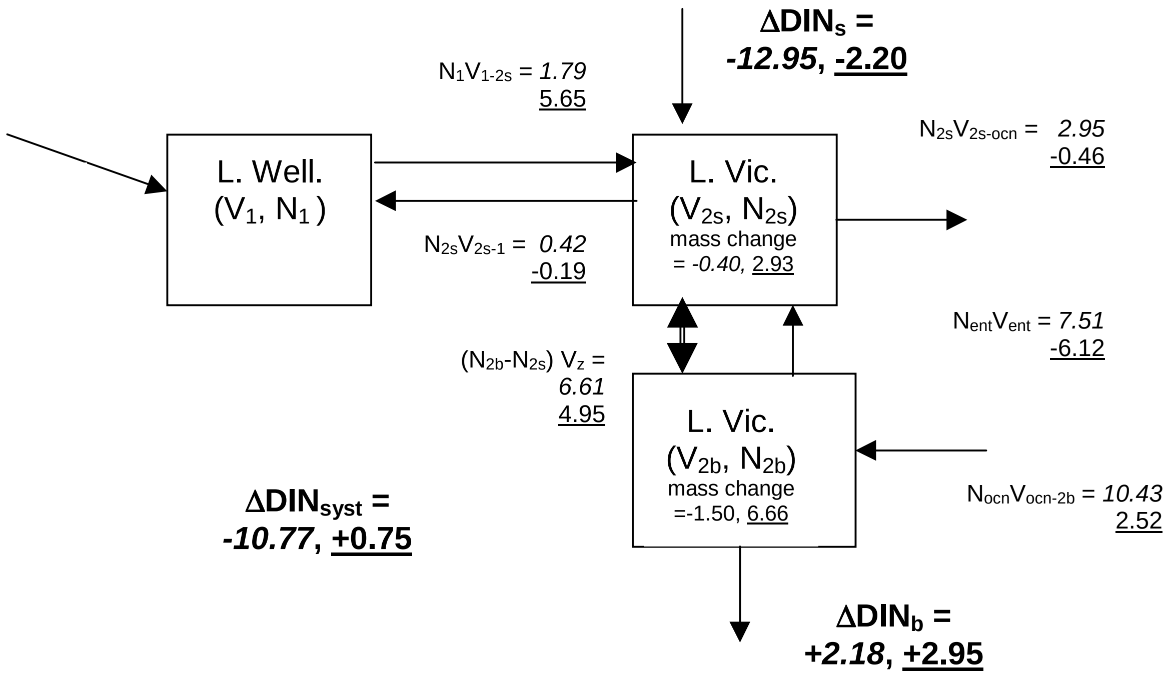


Figure 2.15 DIN budget, Lake Victoria. (DIN concentration of water entering Lake Wellington is not known, although the DIN concentration of water in the lake is known. Therefore the water and salt budgets (Figure 2.13) are used to constrain the exchanging water flow between Lakes Wellington and Victoria. Moreover the changing mass of DIN in Lake Victoria is too large to be treated as 0, so this change is included in the budget.)

Budgets DIP and DIN

DIP Balance

Figure 2.14 displays the P budget for Lake Victoria. In the budgetary analysis, the surface and bottom layers are estimated separately; the surface and bottom data are then combined to establish a whole-lake budget. The calculation uses the concentrations of DIP, DIN and salinity measured at stations 2306, 2311 and 2314 (Figure 2.12). The composition of Lake Wellington inflow water is not known, although the composition of Lake Wellington water is known. Therefore Lake Wellington water is treated as the source water for Victoria. Lake Victoria is a net sink for DIP. Note that the surface layer of the lake is a net sink, while the deep layer is a net source.

DIN Balance

A similar balance is established for DIN in Lake Victoria, with Lake Wellington as the source water (Figure 2.15). Lake Victoria was a sink in 1988-89 and a source in 1989-90. Further, the surface layer was a net sink while the deep layer was a net source in both years.

Stoichiometric Calculations of Aspects of Net System Metabolism

Nitrogen fixation-denitrification (*nfix-denit*) for Lake Victoria was calculated from the difference between the observed and expected ΔDIN for the whole system (surface and bottom), where the expected value is given by ΔDIP x N:P ratio of the organic matter which is reacting (assumed to be phytoplankton with a Redfield N:P ratio of 16:1).

Year 1:

(*nfix-denit*) = $-10.77 \times 10^6 - 16\ (-0.72 \times 10^6) = +0.75 \times 10^6$ mol yr^{-1}

= +0.02 mmol m^{-2} d^{-1}.

Year 2:

(*nfix-denit*) = $0.75 \times 10^6 - 16\ (-0.48 \times 10^6) = 6.93 \times 10^6$ yr^{-1}

= +0.19 mmol m^{-2} d^{-1}.

These calculations suggest that in both years the system was a slight net nitrogen-fixing system.

ΔDIP was also used to calculate the net ecosystem metabolism (*NEM = [p-r]*), as shown below, with the assumption that the reacting organic matter has an approximate Redfield C:P ratio of 106:1.

Year 1:

(p-r) = $-106\ (-0.72 \times 10^6) = +76 \times 10^6$ mol C yr^{-1}

= +2.1 mmol C m^{-2} d^{-1}.

Year 2:
$(p\text{-}r) = -106\ (-0.48 \times 10^6) = +51 \times 10^6$ mol C yr^{-1}

$= +1.4$ mmol C m^{-2} d^{-1}.

The system thus appears to be mildly autotrophic.

Estimates of N and P Fluxes in a Stratified Estuary with Time-Varying Inflows
The system is not necessarily well characterised as being a steady-state system. Therefore a non-steady state analysis was also undertaken. Using the weekly inflow data and interpolating between the fortnightly water column concentration data, daily values for the fluxes of DIN and DIP between Lake Victoria and Lakes Wellington and King were calculated as well as the net internal fluxes. The data were smoothed with a filter that had a cut-off period (half amplitude) of 90 days. The time trend (non-steady state behaviour) of the loading, internal storage, and calculated net fluxes are shown in Figures 2.16 and 2.17. Figures 2.18 and 2.19 illustrate the stoichiometric calculations based on these data.

The important point to note in these analyses is that, while the previous calculations treat the system as if it were at steady state, there is considerable variation in the load and internal nonconservative fluxes for the system. In this case, the mean values obtained by the steady state analysis do not differ greatly from the non-steady state mean values, but the temporal variation is an important characteristic of the system.

Detailed budgetary analysis should take this temporal variability into account (see Webster, Parslow and Smith, Appendix III, for a further analysis of this problem).

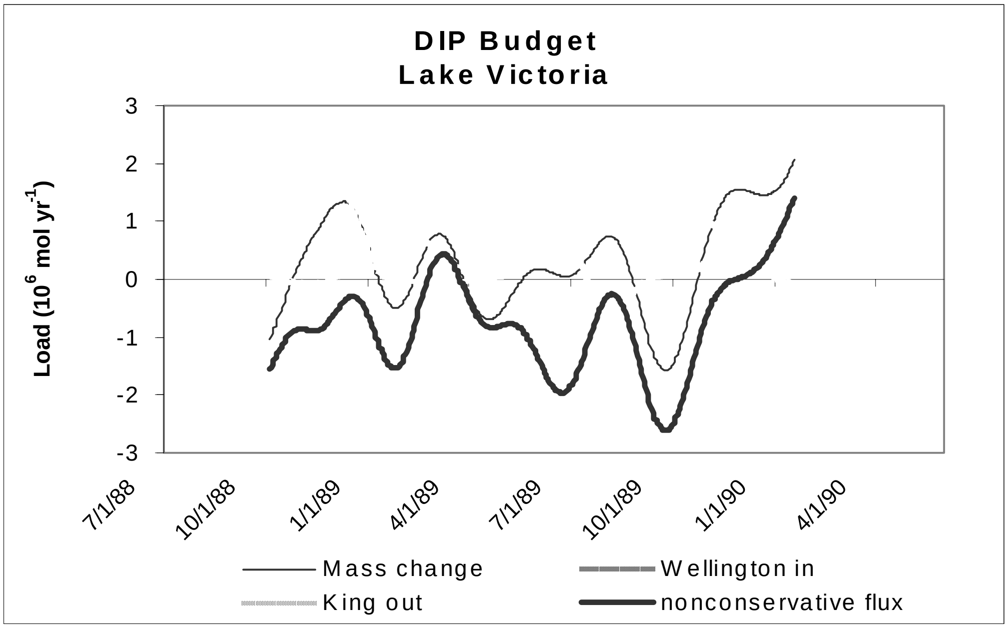


Figure 2.16 Temporal variability (smoothed) in mass of DIP in Lake Victoria, input to it and output from it. (These terms lead to an estimate of the temporal variability in ΔDIP within this system.)

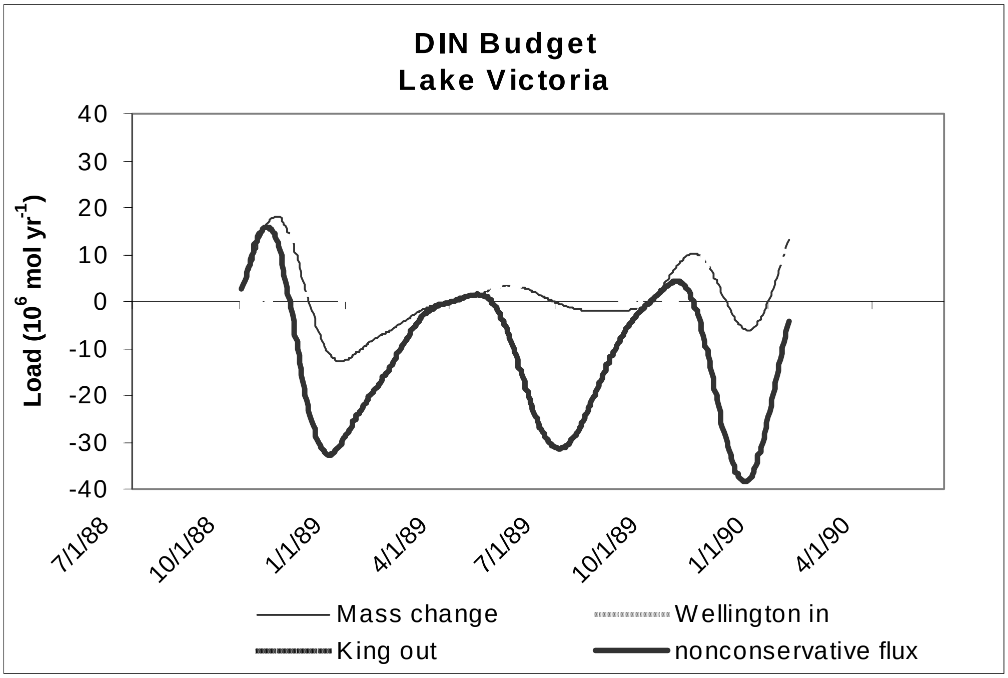


Figure 2.17 Temporal variability (smoothed) in mass of DIN in Lake Victoria, input to it and output from it. (These terms lead to an estimate of the temporal variability in ΔDIN within this system.)

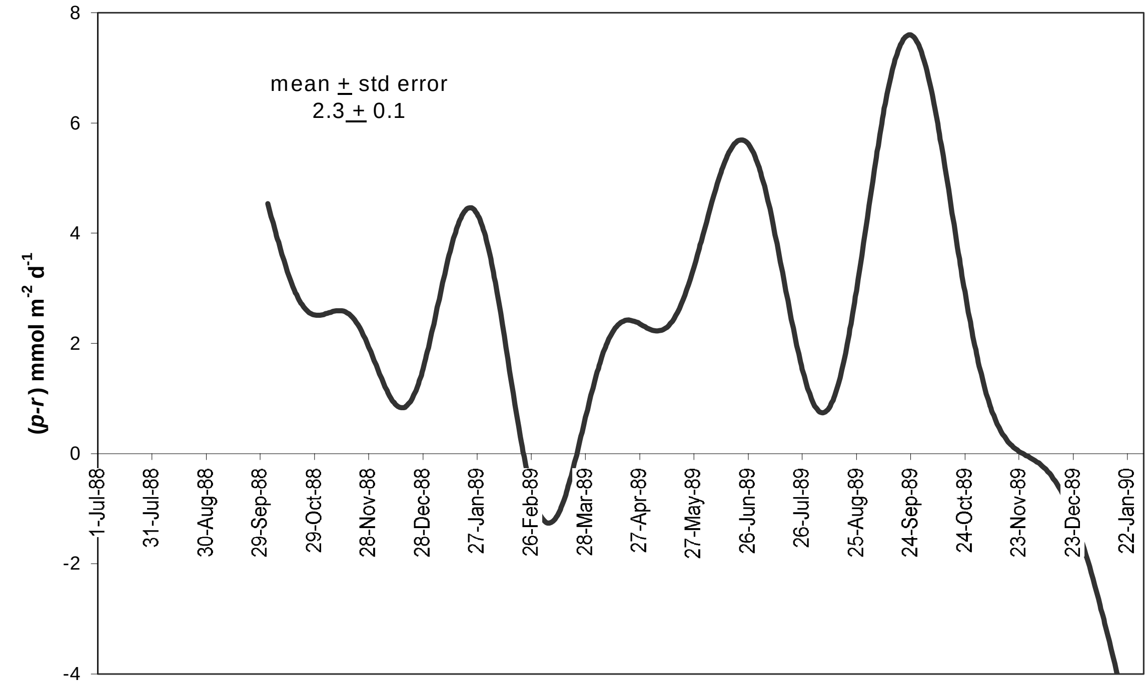


Figure 2.18 Time-varying (*p-r*) for Lake Victoria. (Calculated from Figures 2.16 and 2.17 and scaled per unit area.)

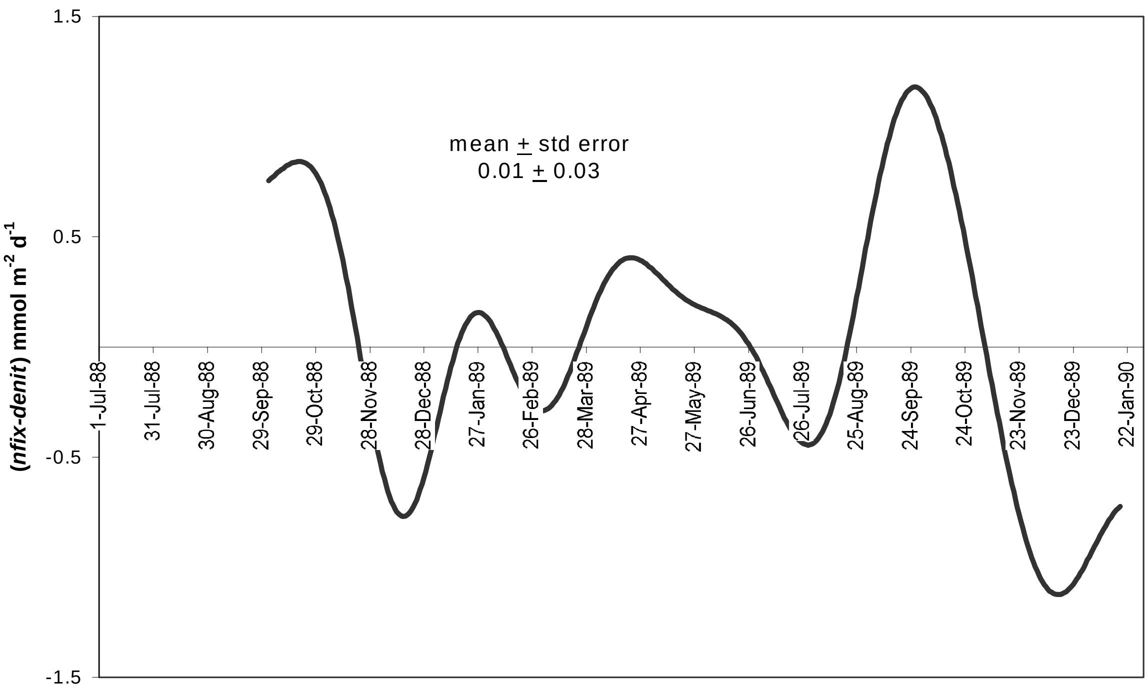


Figure 2.19 Time-varying (*nfix-denit*) for Lake Victoria. (Calculated from Figures 2.16 and 2.17 and scaled per unit area.)

2.5 NITROGEN AND PHOSPHORUS BUDGETS FOR PORT PHILLIP BAY

John Parslow

Study Area Description

Port Phillip Bay (Site No. 25.; 145.00E, 38.00S) is a large (1 900 km^2), shallow coastal embayment in south-eastern Australia (Figure 2.22; see Figure 1.1). The catchment of 9790 km^2 includes the city of Melbourne, with a population of 3 million people. The Bay was the subject of a major environmental study, the Port Phillip Bay Environmental Study (PPBES), conducted from 1992 to 1996 (Harris *et al.* 1996). This study included extensive spatial surveys of water column and sediment nutrient and chlorophyll concentrations, and measurements of primary production, grazing and sediment-water fluxes. Annual nitrogen and phosphorus budgets and a process-based dynamic model of nitrogen cycling through planktonic and benthic systems, were developed as part of the Study (Harris *et al.* 1996, Murray and Parslow 1997). This paper revisits those budgets using LOICZ Biogeochemical Modelling Guidelines.

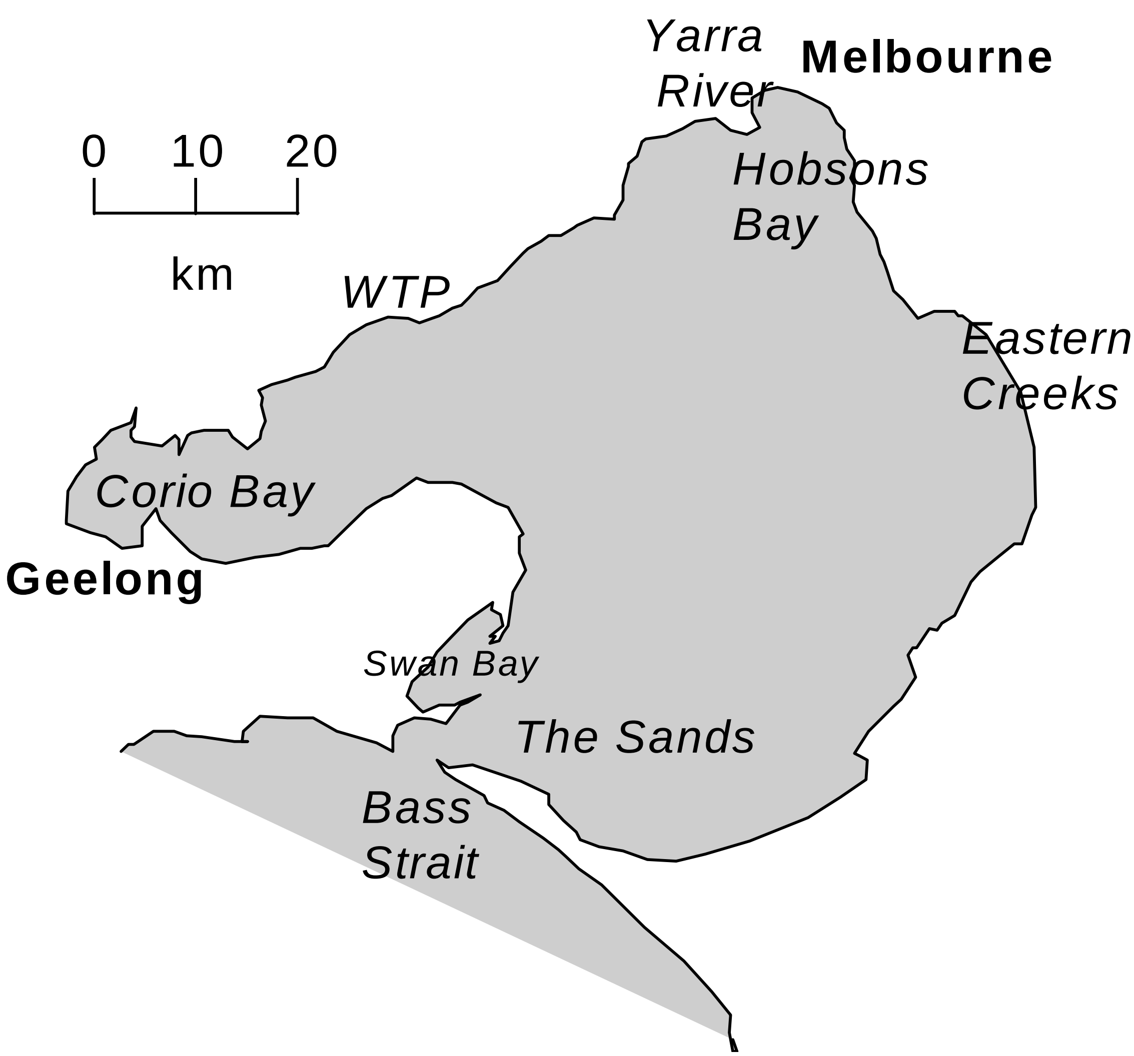


Figure 2.20 Port Phillip Bay, and regions of principal nutrient sources.

Freshwater and salt balance, and flushing rates

Physically, the Bay resembles a broad shallow basin, with a maximum depth of 25 m, mean depth of 13 m, and volume of about 25 x 10^9 m^3. The Bay exchanges with coastal waters (Bass Strait) through a very restricted entrance (the Heads), and across a broad shallow flood tidal delta (the Sands). As part of the PPBES, estimates of flushing rate were obtained in three ways (Walker 1997):

- high resolution measurements of salinity and velocity across the Sands using ADCP and surface radar;
- budgets of salt and freshwater for the entire Bay;
- analysis of a high-resolution 3-D hydrodynamic model and derived transport model.

All three methods agree that the flushing time of the Bay is very long, of order one year. The tidal prism would be sufficient to flush the Bay in about 14 days if tidal exchange were 100% efficient. Tidal flushing is very inefficient because the tidal excursion along the major channels through the Sands is less than the channel length, so that water moves back and forth along the channels without penetrating into the central basin. The instantaneous estimate of flushing time from the high-resolution surveys conducted was 340 days, with an uncertainty of somewhat over 40 days (Walker 1997).

Over the 5 years 1990 to 1994, evaporation was the dominant term in the freshwater budget (86 m^3 s^{-1}), and almost exactly balanced the estimates of combined surface runoff (41 m^3 s^{-1}) and rainfall (42 m^3 s^{-1}) (Harris *et al.* 1996). Uncertainties in the individual estimates means that the net freshwater flux is very uncertain, and cannot be used to compute reliable estimates of flushing rates. Walker (1997) calibrated a transport model so as to reproduce the observed temporal and spatial pattern of salinity in the Bay, and found that this was only possible if evaporation was reduced by 30%, or rainfall estimates increased by 60%. This implies a net freshwater influx V_R = 30 m^3 s^{-1}, or 9.5 x 10^8 m^3 yr^{-1} The mean flushing time for the transport model is about 290 days. The model predicts that effective flushing times vary within the interior of the Bay from about 270 days in the Bay centre adjacent to the Sands to about 315 days in Corio Bay in the far west.

Loads

Loads into the Bay have been estimated using a combination of catchment models and direct estimates (Table 2.14). About half of Melbourne's sewage is treated at the Western Treatment Plant (WTP), which discharges into Port Phillip Bay on the western side (Figure 2.22). The associated nutrient loads have been calculated from weekly measurements of concentration and flow. Loads from the major catchments (Yarra River and Patterson-Mordialloc) have been estimated by fitting empirical catchment models to daily flow time-series and intermittent concentration measurements (Sokolov 1996). Loads from a large number of minor streams and drains are more uncertain, but represent a small fraction (5% or less) of total loads. Atmospheric nitrogen loads have been estimated (via modelling) to be about 71 x 10^6 mol N yr^{-1}, with an uncertainty of about 50%. Estimates of annual loads of ammonium, nitrate, DIN, DIP, organic N, organic P and silicate for the study field period (1993-95) are given in Table 2.14. Silicate was not routinely measured in catchment monitoring programs, and silicate load estimates are based on historical data and considered more uncertain.

Table 2.14 Nutrient loads into Port Phillip Bay in 1993 to 1995 (10^6 mol yr^{-1}).

	Western Treatment Plant	Yarra River	Patterson-Mordialloc	Minor Streams and Drains	Atmosphere	TOTAL
NH_4	154	10	9	3	64	240
NO_3	32	28	20	3	7	89
DIN	186	38	29	5	71	329
DIP	34	11	5	4	0	53
ON	63	61	38	5	0	167
OP	6	3	2	1	0	12
SiO_4	44	47	59	0	0	150

Water Column Concentrations.
During the PPBES field program (1993-95), nutrient concentrations were measured underway along monthly transects designed to sample the spatial gradients within the Bay (Longmore *et al.* 1996). Depth profiles were also measured at a series of 11 stations distributed around the Bay. Except near major streams during high-flow events, the water column was almost always vertically well-mixed.

Table 2.15 Mean concentrations from PPBES spatial surveys (mmol m^{-3}).

	Bay-wide Average	Bay Centre	Heads (Flood)
NH_4	0.64	0.51	0.38
NO_3	0.31	0.13	0.23
DIN	0.95	0.64	0.61
DIP	2.05	1.76	0.24
DON	9.46	8.91	6.12
PTN	1.99	1.59	1.26
ON	11.45	10.50	7.38
TN	12.40	11.14	7.99
DOP	0.21	0.17	0.12
PTP	0.16	0.12	0.09
OP	0.37	0.29	0.21
TP	2.42	2.05	0.45
DOC	1.71	1.56	0.98
POC	0.13	0.09	0.09
OC	1.84	1.65	1.07
SiO_4	4.81	5.12	1.03

Given that the flushing time is long compared with the internal mixing time, we have computed budgets for the Bay as a single compartment, averaged over the 1993-95 field study. However, there are strong gradients within the Bay away from the major point sources, especially in the case of DIN, which is elevated near WTP and the mouth of the Yarra, but very low in the Bay centre. One could estimate export fluxes using the Bay-wide average concentrations and a mean flushing time, but for tracers such as DIN which do not behave conservatively this may over-estimate export, which occurs through exchange between the Bay centre and Bass Strait across the Sands. Instead, we have used Bay centre concentrations and a flushing rate corresponding to the Bay centre. Table 2.15 shows both Bay-wide average concentrations and Bay centre concentrations.

The Bass Strait (boundary) concentration has been estimated by averaging measurements at a station located at the Heads and occupied on flood tide. Mean annual concentration values for this station are given in Table 2.15.

Table 2.16 Loads, export fluxes and net sources of nutrients in Port Phillip Bay. (Net sources are given as both Bay-wide annual fluxes, and average daily fluxes per unit area.)

	Loads (10^6 mol yr^{-1})	Export (10^6 mol yr^{-1})	Δ (10^6 mol yr^{-1})	Δ (mmol m^{-2} d^{-1})
NH_4	211	5	-207	-0.298
NO_3	118	-3	-121	-0.174
DIN	329	2	-327	-0.472
DON		95		
PTN		12		
ON	167	107	-60	-0.087
TN	496	108	-387	-0.559
DIP	53	48	-5	-0.007
DOP		2		
PTP		1		
OP	12	3	-9	-0.013
TP	65	52	-14	-0.020
DOC		19		
POC		0		
OC		20		
SiO_4	150	132	-18	-0.027

N, P and Si Budgets.
The net export fluxes from the Bay to Bass Strait are presented in Table 2.16. These fluxes were calculated using LOICZ Guidelines (Gordon *et al.* 1996) as $V_R \times (C_1+C_2)/2 + V_X \times (C_1-C_2)$, where C_1 is the Bay centre concentration, C_2 is the Heads flood concentration and $V_X = 25 \times 10^9 \times 365/270$ m^3 yr^{-1}. Table 2.16 also presents estimates of net sources (Δ) for the Bay, for those nutrients for which load estimates are available.

Because dissolved and particulate organic forms were not distinguished in inputs, it is not possible to estimate net sources of these forms separately.

Because DIN concentrations are so low in Port Phillip Bay, DIN export is negligible compared with loads. The Bay is a net sink for DIN, and the size of this sink is quite insensitive to flushing rate. The Bay exports organic N, primarily as DON, amounting to 64% of the load of organic N. Refractory DON from the catchment could account for most of this DON export. Of the total N (TN) load of 496 x 10^6 mol yr^{-1}, 22% is exported, and 78% is lost to internal sinks.

By contrast with DIN, most of the DIP load to Port Phillip Bay is exported as DIP. It has long been recognised that Port Phillip Bay is N-limited, and that DIP is present in excess with respect to DIN. It is distributed within the Bay like a conservative tracer. There is a net sink of about 10% of the DIP load, and this estimate is quite sensitive to flushing rate. The data also indicate an additional sink of organic P amounting to about 75% of the organic P load. Of the total P (TP) load of 65 x 10^6 mol yr^{-1}, about 78% is exported, and 22% is lost to internal sinks.

The estimated sink of silicate is 12% of the load, but given the uncertainties in load estimates, export and load can only be said to be in reasonable balance.

Stoichiometry and Internal Sinks.

The internal sink of nitrogen may be attributed either to denitrification, or to burial of organic matter. Phosphorus may be lost through burial of either organic or adsorbed inorganic P. The LOICZ Guidelines assume that burial of adsorbed inorganic P is negligible in coastal waters, and that the DIP sink represents a net production of organic matter. Sediment and water column studies in the Bay suggest that organic matter production is dominated by phytoplankton at close to Redfield ratios. The DIP sink then represents a net community production (*p-r*) of -0.7 mmol C m^{-2} d^{-1}. By comparison, primary production measurements (Harris *et al.* 1996) suggest that net primary production averages over 20 mmol C m^{-2} d^{-1}, so that p:r is less than 1.04. According to the LOICZ interpretation, the internal sink of TP is all due to burial of organic P. At Redfield ratios, this represents a burial rate of 2.12 mmol C m^{-2} d^{-1} (Table 2.17).

According to the LOICZ interpretation, nitrogen fixation-denitrification (*nfix-denit*) is given by: ΔDIN_{obs}-16 x ΔDIP = -254 x 10^6 mol yr^{-1}, or -0.37 mmol m^{-2} d^{-1}. Net denitrification then accounts for 77% of the DIN load, and 51% of the TN load. The organic N export represents 22% of the TN load, and 27% of the TN load (133 x 10^6 mol yr^{-1} or 0.19 mmol N m^{-2} d^{-1}) is buried as organic N (Table 2.17).

Measurements of sediment composition and sediment-water fluxes in Port Phillip Bay suggest a rather different interpretation of the N and P sinks. Benthic chamber measurements showed very high denitrification efficiencies, averaging 70% of the benthic remineralisation (Nicholson *et al.* 1996). The chamber measurements also showed significant adsorption of DIP to sediments (i.e., the flux of DIP out of sediments was lower than predicted from oxygen and carbon fluxes using Redfield stoichiometry).

On average, DIP adsorption in sediments represented about 40% of the DIP expected from benthic remineralisation. This implies a DIP adsorption sink amounting to about 0.036 times the denitrification sink (molar ratio).

Over most of the Bay, particulate C, N and P concentrations in sediments showed little variation with depth over the top 20 cm, and N:P ratios were consistently about 6.3 (by moles). This is much lower than Redfield, and strongly suggests burial of inorganic P, since refractory organic matter buried in sediments would, if anything, be expected to have higher N:P ratios than Redfield. Regardless of the nature of the N and P in sediments, if the sediment N:P ratio is at steady-state, then it should equal the N:P burial ratio. In that case,

$(nfix\text{-}denit) = \Delta TN - 6.3\ \Delta TP$

This yields a value for (*nfix-denit*) of -0.43 mmol $m^{-2}\ d^{-1}$, and for N burial of 0.13 mmol N $m^{-2}\ d^{-1}$ (Table 2.17). The estimated DIP sink due to adsorption according to the benthic flux measurements is then 0.036 times denitrification, or 0.016 mmol P $m^{-2}\ d^{-1}$, leaving a burial rate of organic P of 0.004 mmol $m^{-2}\ d^{-1}$. The net sink of organic P in the Bay is 0.013 mmol $m^{-2}\ d^{-1}$, so a burial rate of 0.004 mmol $m^{-2}\ d^{-1}$ implies a net conversion of the organic P load to DIP amounting to 0.009 mmol $m^{-2}\ d^{-1}$. At Redfield ratios, this implies a net respiration in the Bay of 0.95 mmol $m^{-2}\ d^{-1}$. The further point about these fluxes, of course, is that they are very near 0.

Applying the bulk N:P ratio in sediments to estimate the ratio of TN:TP burial rates is only valid if the sediment composition is at steady-state. This is not necessarily true, because of the long time-scales associated with accumulation of the sediment pools. Rates of sediment accumulation in Port Phillip Bay are very low. The top 20 cm may represent about 1000 years of sediment accumulation, and isotope studies suggest bioturbation mixes this layer on time scales of about 20 years. At the estimated burial rates calculated above, it would take about 110 years to accumulate the observed sediment N and P pools in the top 20 cm. The Bay has been exposed to increasing nutrient loads for the last 150 years following European settlement. It is not clear that the ratio of N:P accumulation has remained constant over this period. However, the benthic chamber studies do provide direct evidence that DIP burial is continuing at a high rate.

Table 2.17 Comparison of N and P fluxes obtained under standard LOICZ interpretation (no DIP adsorption), and from direct measurements of benthic fluxes and sediment composition.

Flux (mmol $m^{-2}\ d^{-1}$)	"Standard" LOICZ Interpretation	Benthic Chamber and Sediment Interpretation
DIP burial	0	0.016
Org P burial	0.020	0.004
Org C (*p-r*)	0.70	-0.95
(*nfix-denit*)	-0.37	-0.43
Org N burial	0.19	0.13

Conclusions.
In freshwater systems, burial of adsorbed inorganic P is often assumed to dominate internal sinks of TP. It is thought to be less important in estuarine and coastal systems, but the Port Phillip Bay example suggests we should be cautious about ignoring it. There is direct evidence from benthic chamber studies in Port Phillip Bay that DIP is adsorbed to sediments, and that ΔDIP should be interpreted primarily as DIP adsorption rather than burial of organic matter. In Port Phillip Bay, denitrification so dominates the N budget that the Bay is apparently net denitrifying regardless of the interpretation of ΔDIP. However, in other systems, ignoring DIP adsorption and burial could lead incorrectly to the conclusion that nitrogen fixation rates are large and dominate the N budget. Where data are sufficient to budget TN and TP, it may be possible to estimate DIP adsorption and burial by measuring the bulk N and P composition of the sediment.

Acknowledgements
The PPBES was funded by Melbourne Water. I acknowledge the efforts of a large number of PPBES participants who collected and analysed the data on which this budget is based.

2.6 DERWENT RIVER ESTUARY, TASMANIA

John Parslow

The Derwent River estuary (Site No. 23.; 147.30E, 42.90S) in Hobart, Tasmania (Figure 2.21; see Figure 1.1) is a drowned river valley, draining a large catchment, with (by Australian standards) consistently high runoff. Much of the catchment is undisturbed, although there is agricultural development in the downstream parts of the catchment. The city of Hobart sits astride the Derwent, and the estuary receives nutrient loads from a large number of small wastewater treatment plants (WTPs), and some industrial sources.

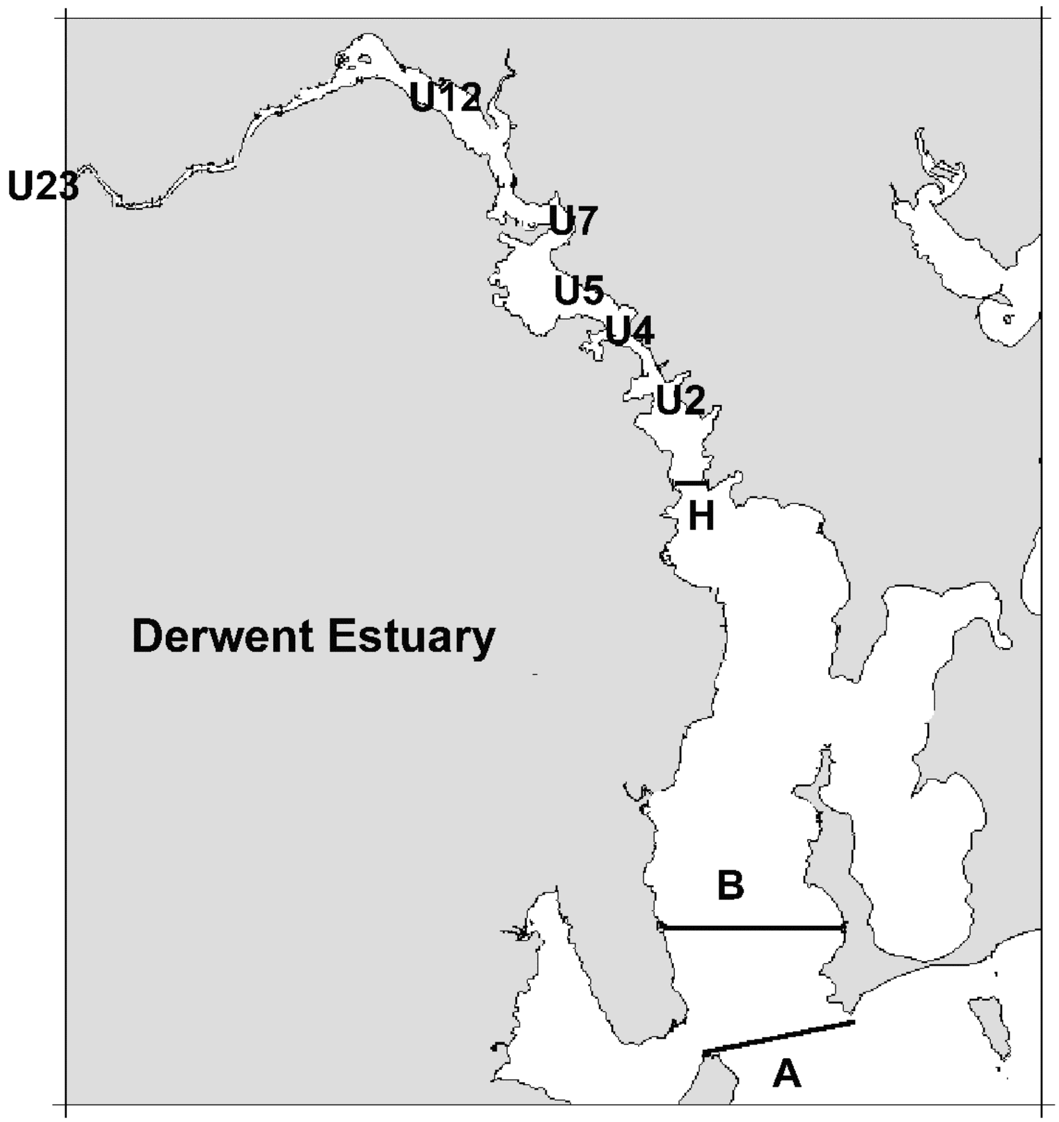


Figure 2.21 Derwent River estuary, showing sampling stations in CSIRO surveys.

The Derwent River estuary is relatively deep and, given the high runoff and low tidal amplitude, represents a classic example of a Pritchard-type salt-wedge estuary. The estuary can be divided length-wise into three sections (Figure 2.21). The upper estuary from New Norfolk down to just above Bridgewater, a distance of about 13 kms, is relatively narrow, and dominated by river flow. The middle estuary, from just above Bridgewater down to Hobart, is of the order of 1 km wide, and includes a number of shallow side embayments. The lower estuary, from Hobart to the Iron Pot, is much broader (order of 5 km) and generally marine-dominated. One might expect diffusive exchanges to be more important in the lower estuary.

The Derwent River discharge is by far the dominant freshwater input into the estuary. The river is dammed and used to generate hydro-electricity, so the discharge is strongly regulated. While flow may drop to about 20 $m^3 s^{-1}$ for brief periods, it is typically greater than 50 $m^3 s^{-1}$, and peak discharge can reach 300 $m^3 s^{-1}$ (Coughanowr 1995).

Coughanowr (1995, 1997) provides estimates of average DIN and DIP loads from the WTPs and industrial sources along the Derwent. The point source loads in 1996 are summarised in Table 2.18. These loads are assumed to be constant within years.

Table 2.18 DIN and DIP loads from point sources (WTP and industrial) along the Derwent estuary.

LOCATION	DIN LOAD (mol d^{-1})	DIP LOAD (mol d^{-1})
U23	2 661	363
U21	12 369	1 669
U11	9 481	1 249
U6	1 536	502
U4	18 440	2 776
U3	7 617	857
U2	13 776	1 808
H	8 967	1 245
F	1 976	232
D	7 314	938
B	2 740	0

The Tasmanian Department of Environment and Land Management (DELM) conducted surveys of the Derwent River estuary in 1994 and 1996. Physical, chemical and biological properties were sampled at a series of locations along the length of the estuary, with some additional sampling in the side embayments. DELM has made available data collected in May 1994, and January, February, April and July 1996. Summaries of these data have been published in two technical reports (Coughanowr 1995, 1997), which also identify the principal industrial and WTP point sources, and provide estimates of nitrogen loads from these sources. (These reports also provide an excellent general description of the estuary and catchment.)

Biogeochemical Budgets for May 1994

In May 1994, salinity, DIN, DIP and TP were sampled top and bottom at 38 stations in the estuary, corresponding to a subset of the standard station grid shown in Figure 2.21. In the lower estuary, samples were collected at a number of stations on transects across the estuary, and these have been averaged to produce a 1-D set of top and bottom layer concentrations C_U and C_B at 25 locations along the estuary.

The estuary is budgeted by assuming that there is a classic 2-layer Pritchard-type circulation, with downstream flow in the top layer, upstream flow in the bottom layer, and vertical entrainment and mixing between top and bottom layers. We start by considering the net downstream flux of freshwater, salt and other tracers in the estuary. This is equivalent to the two-layer box models as illustrated in the LOICZ Guidelines (Gordon *et al.* 1996).

At any point along the estuary, if V_U is the downstream transport ($m^3\ s^{-1}$) in the upper layer, V_B is the upstream transport in the lower layer, and V_Q is river runoff, then:

$$V_U = V_B + V_Q$$

The net downstream salt flux, given by $V_U.S_U$-$V_B.S_B$, must be zero, and it follows that:

$$V_B = V_Q S_U/(S_B-S_U)$$

The net downstream flux of any other tracer C at a given point along the estuary is then given by:

$$F = V_U C_U - C_B$$

Given the river runoff, V_Q, we have computed the upstream transport, $V_{B,}$ and the net downstream flux of DIN, DIP and TP at each of the 25 locations where these tracers, plus salinity, were measured in top and bottom layers. The transport V_B is plotted versus distance downstream in Figure 2.22, and the net downstream flux of DIN and DIP (mol d^{-1}) are plotted versus distance downstream in Figure 2.23.

In an idealised steady-state 2-layer flow, the salinity in the top and bottom layers should increase monotonically downstream. The observed salinity in the top layer does increase monotonically, but the observed salinity in the bottom layer does not, due possibly to variations in both bottom depth and sampling depth along estuary, and to transverse variation or tidal aliasing in the lower estuary. Effects of this can be seen in Figure 2.22 where the transport, V_B, which should increase monotonically down the estuary, undergoes local fluctuations.

The estimated net downstream flux of DIN and DIP also fluctuates from point to point, in some cases changing sign. The DIN flux is low and positive and declines smoothly in the upper reaches of the estuary, and then increases but fluctuates substantially in the central

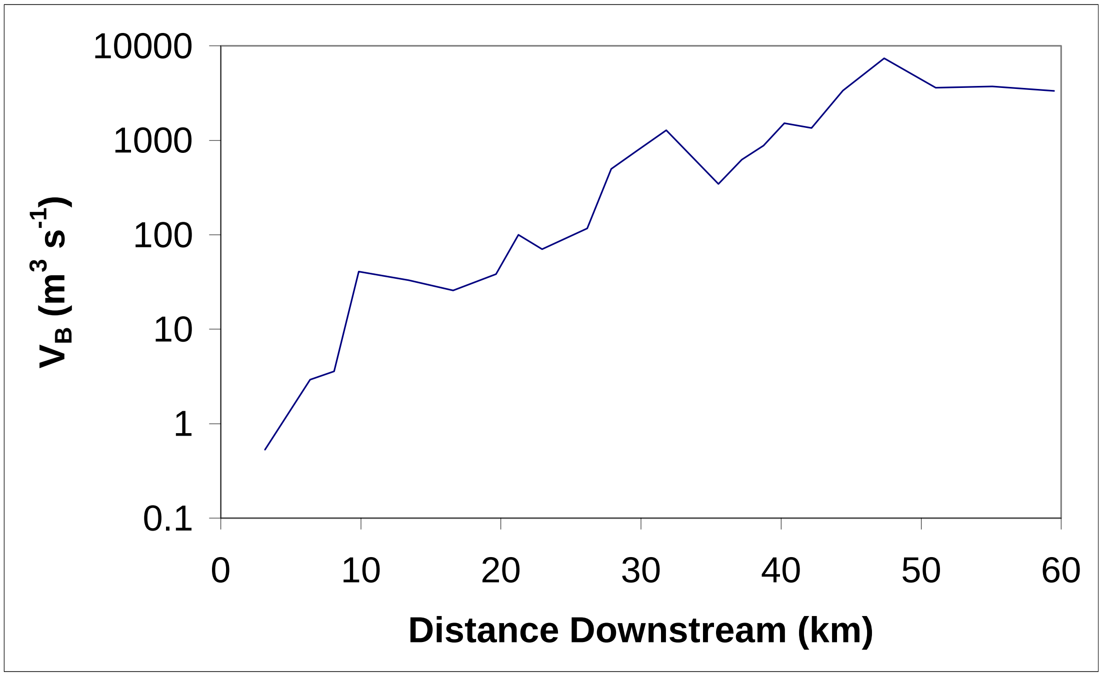


Figure 2.22 Calculated bottom transport V_B versus distance downstream, Derwent River estuary.

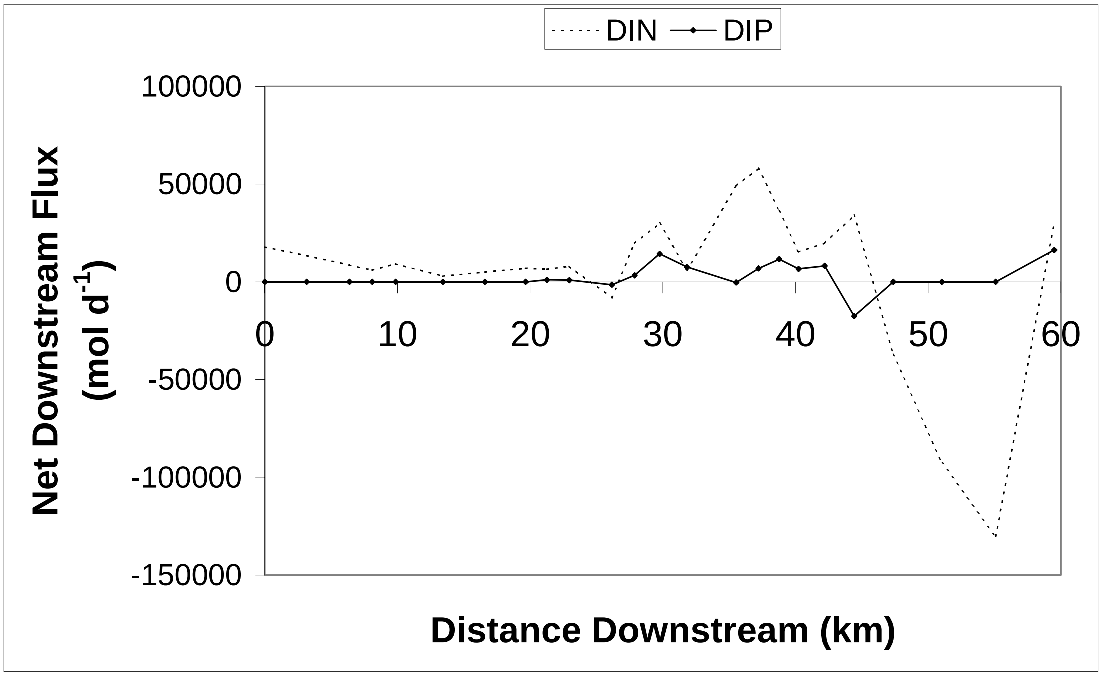


Figure 2.23 Net downstream flux of DIN and DIP versus distance downstream, Derwent River estuary.

reaches. The net flux depends strongly on the difference between upper and lower layer concentrations. In the central region, concentrations of DIN are elevated in both layers, with a small and variable difference between the layers. In the lower estuary DIN concentrations are slightly higher in the bottom layer and this, combined with the very large estimated transport, leads to a large estimated net flux of DIN upstream (i.e., into the estuary). However, this flux is very uncertain and in fact the concentration difference between lower and upper layers and the sign of the flux reverses at the mouth of the estuary.

In the case of DIP, the measured concentrations were below detection in the upper and lower reaches of the estuary in both top and bottom layers, resulting in zero flux estimates. Estimated fluxes were positive but variable in the middle reaches and at the mouth.

In principle, one could estimate the net source of DIN and DIP between each pair of observations by finding the difference between the net downstream fluxes. However, given the variability along the estuary, these local source-sink estimates are very noisy and difficult to interpret. Instead, we took the point source loads and locations in Table 2.18, and added these to the river load to give cumulative loads as a function of distance downstream. We then subtracted the cumulative load upstream of each sample point from the net downstream flux at that point, to estimate the net internal source upstream of each sample point. This is plotted versus distance downstream in Figure 2.24.

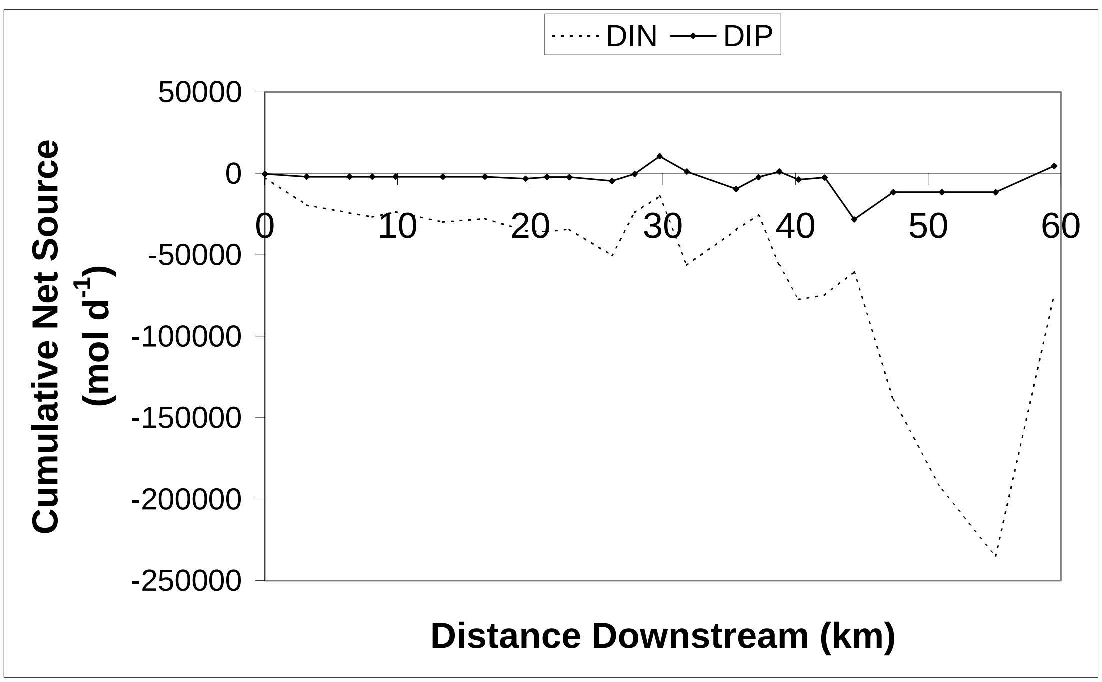


Figure 2.24 Calculated net source of DIN and DIP upstream of each sampling point versus distance downstream, Derwent River estuary.

On the basis of Figure 2.24, the estuary can be divided into three regions corresponding roughly to the upper, middle and lower estuaries. There is always a net sink of DIN throughout the estuary, i.e., the net downstream transport at each point is always less than the load upstream of that point. In the upper estuary, the net DIN sink increases smoothly with distance downstream. In the middle estuary, where most of the point source loads occur, the sink appears to continue to increase, but fluctuates substantially. In the lower estuary, there is apparently a very large sink but this reverses abruptly at the mouth.

There is a small net sink of DIP in the upper estuary, and this increases in magnitude but fluctuates in sign through the middle estuary. Again, there appears to be a large sink in the lower estuary, but this reverses to become a source at the mouth.

On the basis of Figure 2.24, we have computed regional budgets for three subregions: the upper, middle and lower estuary. These budgets are shown in Tables 2.19 and 2.20. In the upper estuary, DIN export is much less than load, so there is a large sink of DIN both in absolute terms and on an areal basis. A substantial fraction of the DIP load and the TP load is also lost to an internal sink. If the DIP sink is interpreted as organic carbon production, then there is a very high net rate of internal production (about 20 mmol m^{-2} d^{-1}), and nitrogen fixation-denitrification is close to zero. On the other hand, if we interpret the DIP loss as due to adsorption to sediments, then there is a very high rate of denitrification. Either explanation is plausible, as this reach of the estuary is relatively fresh (salinity <10). There is little phytoplankton production, but in the lower reaches, there are shallow bays with extensive beds of macrophytes that could result in considerable net organic production. The upper estuary has also been subject to large loads of organic carbon from the ANM paper mill. These have produced anoxic mats that could lead to high rates of denitrification.

	mol d^{-1}	Upper	Middle	Lower (B)	Lower (A)
DIN	Load	42 284	50 335	12 030	12 030
	Export	6 418	29 923	-166 723	-6 438
	Δ	-35 866	-20 412	-178 753	-18 468
DIP	Load	3 281	7 189	1 170	1 170
	Export	1 002	10 571	-11 573	4 593
	Δ	-2 280	3 382	-12 742	3 423
TP	Load	5 065	7 189	1 170	1 170
	Export	2 398	-4 164	6 895	-31 516
	Δ	-2 667	-11 353	5 725	-32 686

Table 2.19 Fluxes of DIN, DIP and TP for upper, middle and lower estuary, Derwent River. (Upper estuary extends downstream to sampling point U10, middle estuary to H on Figure 2.21. Two budgets are presented for the lower estuary, upstream of transects B and A respectively. We consider the lower estuary calculation at transect B to be the more reliable.)

	mmol m^{-2} d^{-1}	Upper	Middle	Lower (B)	Lower (A)
DIN	Load	3.46	2.12	0.10	0.09
	Δ	-2.94	-0.86	-1.48	-0.14
DIP	Load	0.27	0.30	0.01	0.01
	Δ	-0.19	0.14	-0.11	0.03
TP	Load	0.41	0.30	0.01	0.01
	Δ	-0.22	-0.48	0.05	-0.24
	(*p-r*)	19.80	-15.07	11.22	-2.68
	(*nfix-denit*)	0.05	-3.13	0.21	-0.54

Table 2.20 Fluxes of DIN, DIP and TP for upper, middle and lower estuary, Derwent River, as defined in Table 2. (Net ecosystem production (*p-r*) and nitrogen fixation-denitrification *(nfix-denit*) are calculated as suggested by the LOICZ methodology, assuming Redfield ratios.)

In the middle estuary, DIN export is less than loads and there is again a substantial internal DIN sink, although this is much smaller on a per area basis. There is apparently a DIP source in this section, which could be due to release of P from bottom sediments. However, the magnitude of this source is uncertain, given long-stream variation in DIP transport. If one interprets the DIP source as being due to breakdown of organic matter at Redfield ratios, this is a region of net respiration and very high rates of denitrification per unit area. If the DIP source is inorganic particulate P, then this is a region of moderate denitrification.

Budgets in the lower estuary vary considerably, depending on whether the marine boundary is set at sampling transect A or B. If the boundary is set at B, there is a large flux of DIN and DIP into the estuary from offshore, and a large internal sink of DIN and DIP. If this is converted to organic matter at Redfield ratios, it implies high net ecosystem production in the lower box. The DIN and DIP influx are close to Redfield balance, and the calculated net rate of nitrogen fixation is small compared with the DIN sink, and insignificant given the uncertainties in fluxes. If the boundary is set at A, the marine influx disappears, and the internal sink of DIN in the lower estuary becomes quite small. The net DIP source is essentially zero. It is unclear whether this variation represents a real characteristic of the system or uncertain results.

It is possible by budgeting salt and freshwater to estimate the fluxes due to vertical entrainment and vertical mixing in each section of the estuary, and therefore to compute the net sources of DIN and DIP in top and bottom layers separately. We have carried out these computations both between each pair of sampling points and for the broad regions (upper, middle and lower) just discussed. However, these computations are even more sensitive to small errors in observations, and both sets of calculations give very large compensating sources and sinks in top and bottom layers. We have decided not to present them here until the error terms in the budgets are better understood.

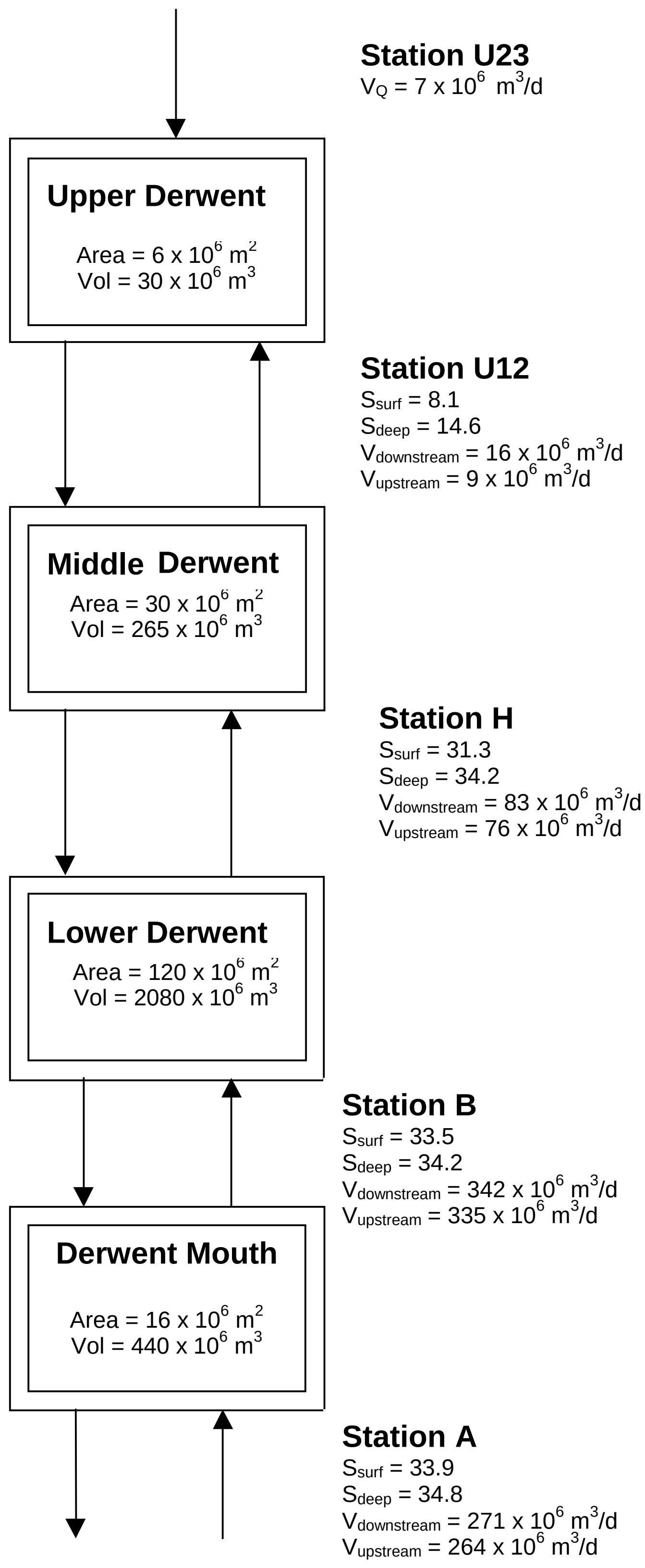


Figure 2.25 Water and salt budgets, Derwent River estuary.

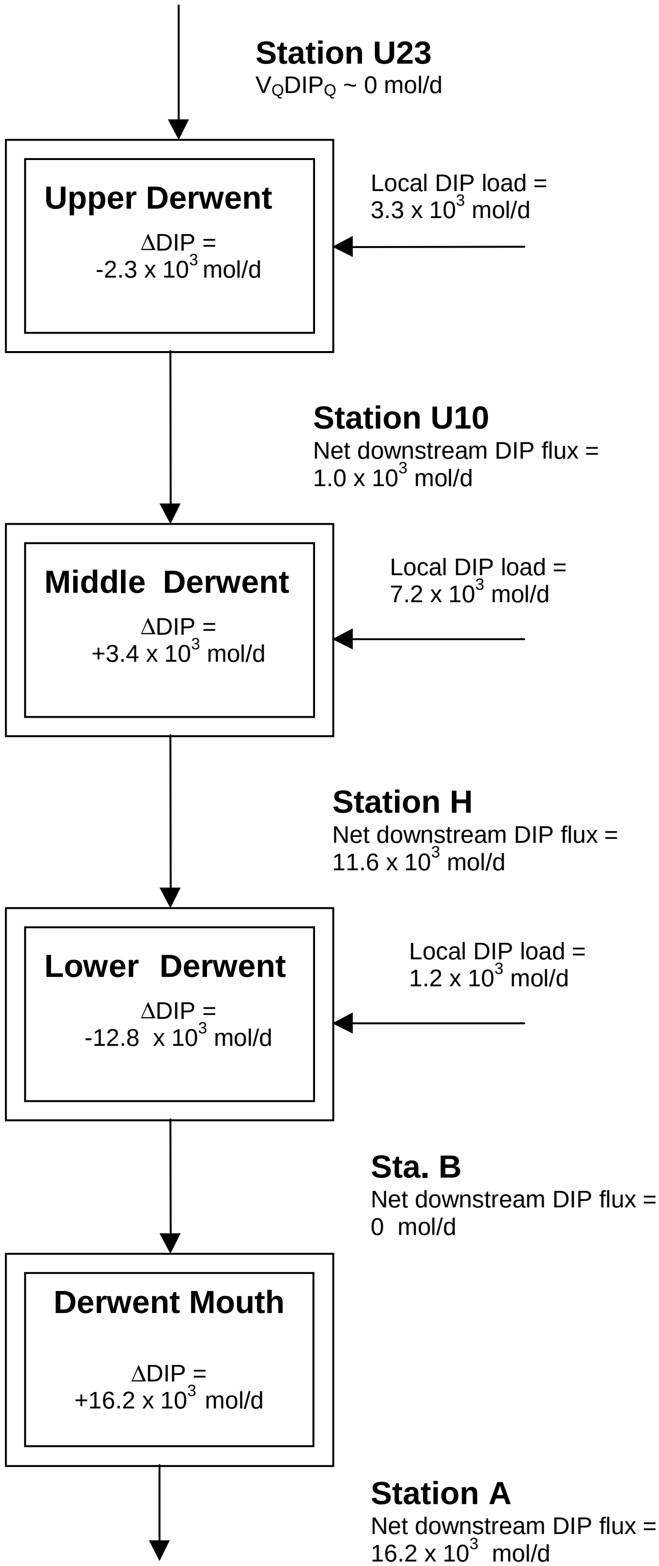


Figure 2.26 DIP budget, Derwent River estuary.

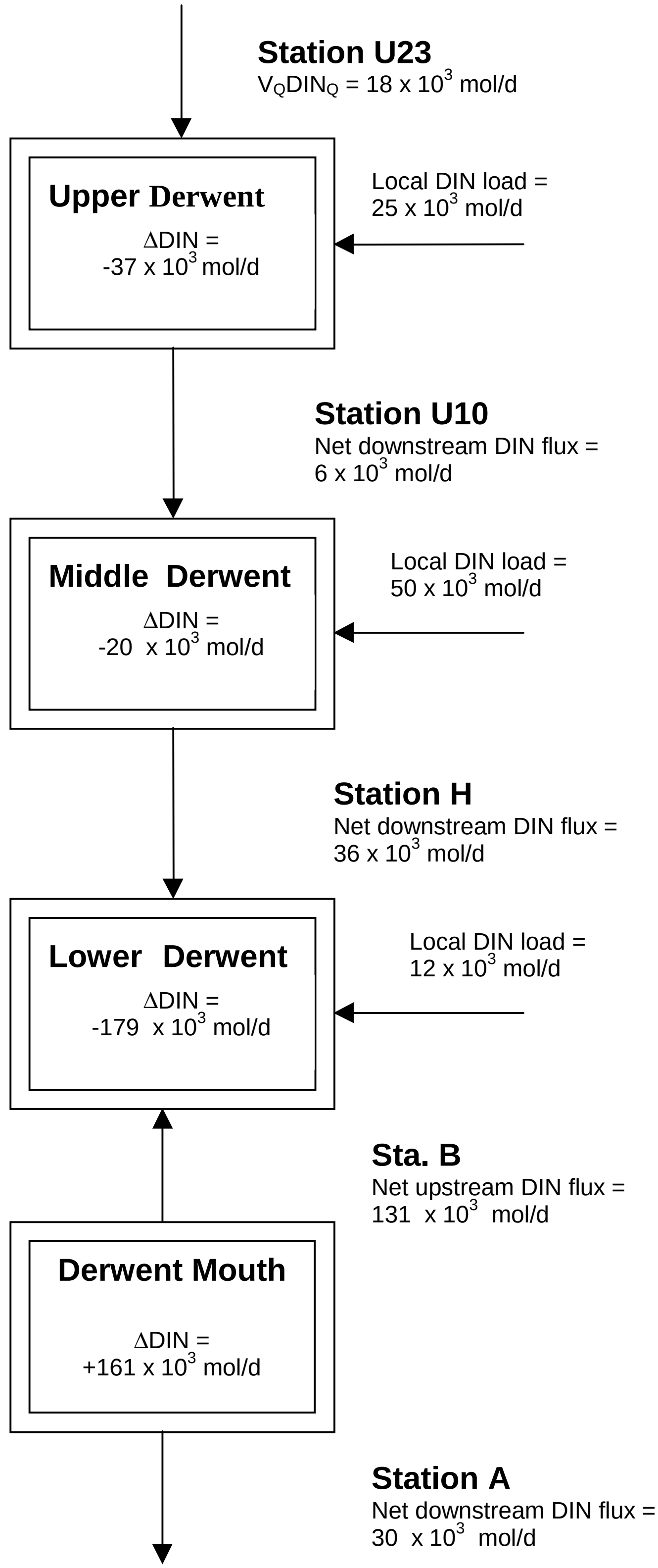


Figure 2.27 DIN budget, Derwent River estuary.

Figures 2.25-2.27 convert the calculations presented above into LOICZ-style box diagrams for the various sectors. The water exchange time for the whole estuary (system volume divided by the downstream water flux; Figure 2.27) is approximately 10 days. While the division into estuary segments is desirable to estimate the fluxes precisely, the water exchange times in each sector are very short (2-4 days for the upper and middle sectors). Therefore, we conclude that the overall system budgets based on accumulation of the data for the whole estuary provide the most robust estimates of system performance. The analysis by subsector is presented for illustrative purposes only.

ΔDIP for the entire estuary totals +4,500 mol/day, equivalent to +0.03 mmol m^{-2} d^{-1}. This is converted to an estimate of (*p-r*) by multiplying by -106, on the assumption that organic metabolism is dominated by plankton; (*p-r*) is estimated by the LOICZ procedures to be -3 mmol m^{-2} d^{-1}. According to this interpretation, the system is slightly net heterotrophic, although the difference from 0 is probably insignificant. ΔDIN for the system is -75,000 mol/d, or -0.4 mmol m^{-2} d^{-1}; (*nfix-denit*) is calculated as ΔDIN_{obs} - ΔDIN_{exp}, or -0.9 mmol m^{-2} d^{-1}.

Budgets for January, February, April and July 1996

Data are not available for a full annual cycle. However, DELM conducted some further surveys in 1996. On these surveys, DIN, DIP, TN and TP were measured. Unfortunately the bottom layer was only sampled on these surveys at one or two stations in mid-estuary. Consequently it has only been possible to compute net downstream fluxes at these stations and net budgets for regions of the estuary upstream of these stations. The budgets are summarised in Table 2.21. We present these budgets to give a sense of the seasonal variation in the non-conservative fluxes, with the caveat given above that we do not place a great deal of confidence in either the non-conservative performance of the budgets or the stoichiometric interpretations based on the upper estuary, due to very short water residence time and relatively low salinity of this portion of the estuary. Because slightly different stations are used for each calculation, the nonconservative fluxes are simply presented as normalised per unit area.

Export of DIN is consistently much less than the DIN load, and in February and April there is a weak net flux of DIN upstream at one or both stations. In July there is a large net upstream flux of DIN, driven by elevated nitrate due to winter mixing on the adjacent continental shelf. Consequently, the budgets consistently show a large internal sink of DIN, ranging from 1.4 to 5.6 mmol m^{-2} d^{-1}.

The net export of TN is always positive and, except in January, represents a substantial fraction of the load. There is still a substantial sink of TN within the estuary, ranging from 1.9 to 5.8 mmol m^{-2} d^{-1}, except in July, when this sink is only 0.8 mmol m^{-2} d^{-1}.

DIP behaves much like DIN – the export is always less than load, and in some months, especially July, there is a significant influx of DIP from the ocean. The net sink of DIP within the estuary ranges from 0.14 to 0.8 mmol m^{-2} d^{-1}. If this is assumed converted to organic matter at Redfield ratios, it represents extremely high rates of net community production, up to 87 mmol C m^{-2} d^{-1} in July. If real, this is unlikely to represent

Table 2.21 Calculated budgets for region of Derwent River estuary upstream of nominated sampling stations, in 1996.

	DIN	TN	DIP	TP	(*nfix-denit*)	(*p-r*)
JANUARY (R = 60 m^3 s^{-1})						
U5 (V_B=241 m^3 s^{-1})						
Export (mol d^{-1})	1 550	3 753	104	-179		
Load (mol d^{-1})	34 872	120 717	4 145	5 232		
Δ (mol d^{-1})	-33 323	-116 964	-4 041	-5 411	31 334	428 350
Δ (mmol m^{-2} d^{-1})	-1.37	-4.79	-0.17	-0.22	1.28	17.55
FEBRUARY (R = 100 m^3 s^{-1})						
U7 (V_B=27 m^3 s^{-1})						
Export (mol d^{-1})	-2 900	119 058	-499	2 242		
Load (mol d^{-1})	36 237	163 368	3 839	5 232		
Δ (mol d^{-1})	-39 137	-44 311	-4 337	-2 990	30 261	459 762
Δ (mmol m^{-2} d^{-1})	-2.42	-2.74	-0.27	-0.19	1.87	28.45
U4 (V_B=81 m^3 s^{-1})						
Export (mol d^{-1})	-9 007	113 371	-1 265	1 959		
Load (mol d^{-1})	56 222	183 353	7 124	8 517		
Δ (mol d^{-1})	-65 229	-69 982	-8 389	-6 559	69 000	889 267
Δ (mmol m^{-2} d^{-1})	-2.43	-2.60	-0.31	-0.24	2.57	33.10
APRIL (R = 200 m^3 s^{-1})						
U5 (V_B=106 m^3 s^{-1})						
Export (mol d^{-1})	2 496	180 095	1 477	6 034		
Load (mol d^{-1})	92 699	321 042	4 898	6 570		
Δ (mol d^{-1})	-90 202	-140 946	-3 420	-536	-35 474	362 572
Δ (mmol m^{-2} d^{-1})	-3.69	-5.77	-0.14	-0.02	-1.45	14.85
U2 (V_B=329 m^3 s^{-1})						
Export (mol d^{-1})	21 490	286 718	-8 366	2 346		
Load (mol d^{-1})	132 531	360 874	10 339	12 012		
Δ (mol d^{-1})	-111041	-74 156	-18 705	-9 665	188 240	1 982 741
Δ (mmol m^{-2} d^{-1})	-3.55	-2.37	-0.60	-0.31	6.02	63.38
JULY R = 90 m^3 s^{-1}						
U5 (V_B=175 m^3 s^{-1})						
Export (mol d^{-1})	-46 503	71 985	-9 922	1 405		
Load (mol d^{-1})	51 597	81 590	4 285	4 786		
Δ (mol d^{-1})	-98 100	-9 605	-14 206	-3 381	129 203	1 505 883
Δ (mmol m^{-2} d^{-1})	-4.02	-0.39	-0.58	-0.14	5.29	61.69
U2 (V_B=455 m^3 s^{-1})						
Export (mol d^{-1})	-83 034	95 469	-16 021	-3 127		
LOAD (mol d^{-1})	91 430	121 423	9 726	10 228		
Δ (mol d^{-1})	-174 464	-25 953	-25 748	-13 355	237 497	2 729 243
Δ (mmol m^{-2} d^{-1})	-5.58	-0.83	-0.82	-0.43	7.59	87.24

phytoplankton production, as phytoplankton biomass is low and phytoplankton production appears to be heavily light-limited in winter.

Except for station U5 in April, the DIP sink times Redfield N:P ratio always exceeds the DIN sink, so that if the DIP sink is interpreted as being due to organic matter production, it implies net N-fixation. Very high rates of N-fixation are estimated in this way in April and July. This interpretation does not seem plausible, as it implies high rates of N-fixation at a time when light and temperatures are low, and DIN concentrations in the water column are elevated. This situation may represent burial of P as inorganic material, or it may simply be an artifact of budgeting a region with an excessively short exchange time.

The export flux of TP is generally greater than the export flux of DIP, except in January. In July, when there is a large influx of DIP past both stations, there is a significant efflux of TP past the upstream station, and only a small influx past the downstream station. Internal TP sinks are significantly smaller than DIP sinks, so that a significant portion of the DIP sink is exported either as adsorbed inorganic particulate P or as organic P.

Conclusions

The May 1994 budget showed considerable local variation in downstream fluxes in the middle estuary. This suggests that the calculated budgets in 1996, based on one or two stations in the middle estuary, are subject to significant errors, which could be exaggerated in the calculation of derived quantities such as (*nfix-denit*). This is particularly a problem, given the very short water residence time in the upper portion of the estuary. On the other hand, the 1994 and 1996 budgets consistently show large sinks of DIN and DIP in the upper and middle estuary, both in summer, when point source loads are the dominant inputs, and in winter, when influx from the adjacent continental shelf dominates. These sinks must be attributed either to high rates of organic matter production and burial/export, or to high rates of DIP adsorption and denitrification. Light attenuation is high in the Derwent River estuary, and phytoplankton biomass and production are strongly light-limited. High net rates of organic matter production may be associated with benthic macrophytes in shallow side embayments in the middle estuary. High rates of denitrification may be facilitated by historically large loads of organic carbon from the ANM paper mill.

2.7 COCKBURN SOUND, WESTERN AUSTRALIA

A. W. Chiffings

Cockburn Sound (Site No.28.; 115.80E, 32.20S) is the largest of several coastal embayments immediately south of the port of Fremantle on the Western Australian coast (Figure 2.28; see Figure 1.1). The Sound has a 45-year history of ecosystem change as a result of perturbations from industrial development over this time. Scientific studies have been undertaken on the Sound since the late 1970s and much of this effort has been focused on gaining an understanding of key physical, biochemical and biological processes. As a result of these studies it is thought now to be possible to look at long-term changes in the Sound from a systems point of view, particularly in terms of nutrient materials flux using the LOICZ approach (Gordon *et al*., 1996).

The initial models presented here have been prepared using data obtained from a study (1977 – 1983) that gained a quantitative understanding of the relative importance of physical exchange and mixing processes, nutrient additions and sediment recycling to phytoplankton-nutrient interrelations in the nearshore coastal waters of the region.

As time permits, models based on the subsequent studies through to the most recent studies (1994) are being prepared so that comparisons can be made of the impacts of changes in both ecosystem functioning and nutrient loading regimes on overall behaviour of the system.

Study Area Description

Cockburn Sound is one of four coastal basins formed from the flooding of a depression between Pleistocene aeolianite ridges running north-south, and the subsequent deposition of east-west Holocene banks (Churchill 1959, France 1978). Warnbro Sound, the southern embayment, is separated from Cockburn Sound by a Holocene bank overlain by a series of emergent beach ridges (Fairbridge 1950, Wood 1983). The banks separating Cockburn Sound from Owen Anchorage (Parmelia Bank), and Owen Anchorage from Gage Roads (Success Bank), are submerged shallows covered to a considerable extent by seagrasses (Cambridge 1975). Gage Roads, the northern-most basin, is open to the north. The Swan-Canning River system enters Gage Roads at the port of Fremantle (Figure 2.28), with the Peel-Harvey estuary and riverine system entering the coast some 16 kms to the south of Warnbro Sound.

The eroded aeolianite ridge on the western side of the embayments forms a chain of submerged and partly exposed reefs, rocky outcrops and islands of various size. The largest of the islands, Garden Island, makes up the predominant part of the western boundary of Cockburn Sound.

Cockburn Sound has a surface area of 114 km^2 and a mean depth of 12 m. It is almost entirely made up of a deep basin (16-20 m deep) with shallow margins that were once covered by seagrass. These margins extend out into a broad shelf (approximately 3 km wide and 4-7 m deep) between James Point and Woodman Point along the eastern side of the Sound. The other dominant feature is an extensive shallow sand flat in the south

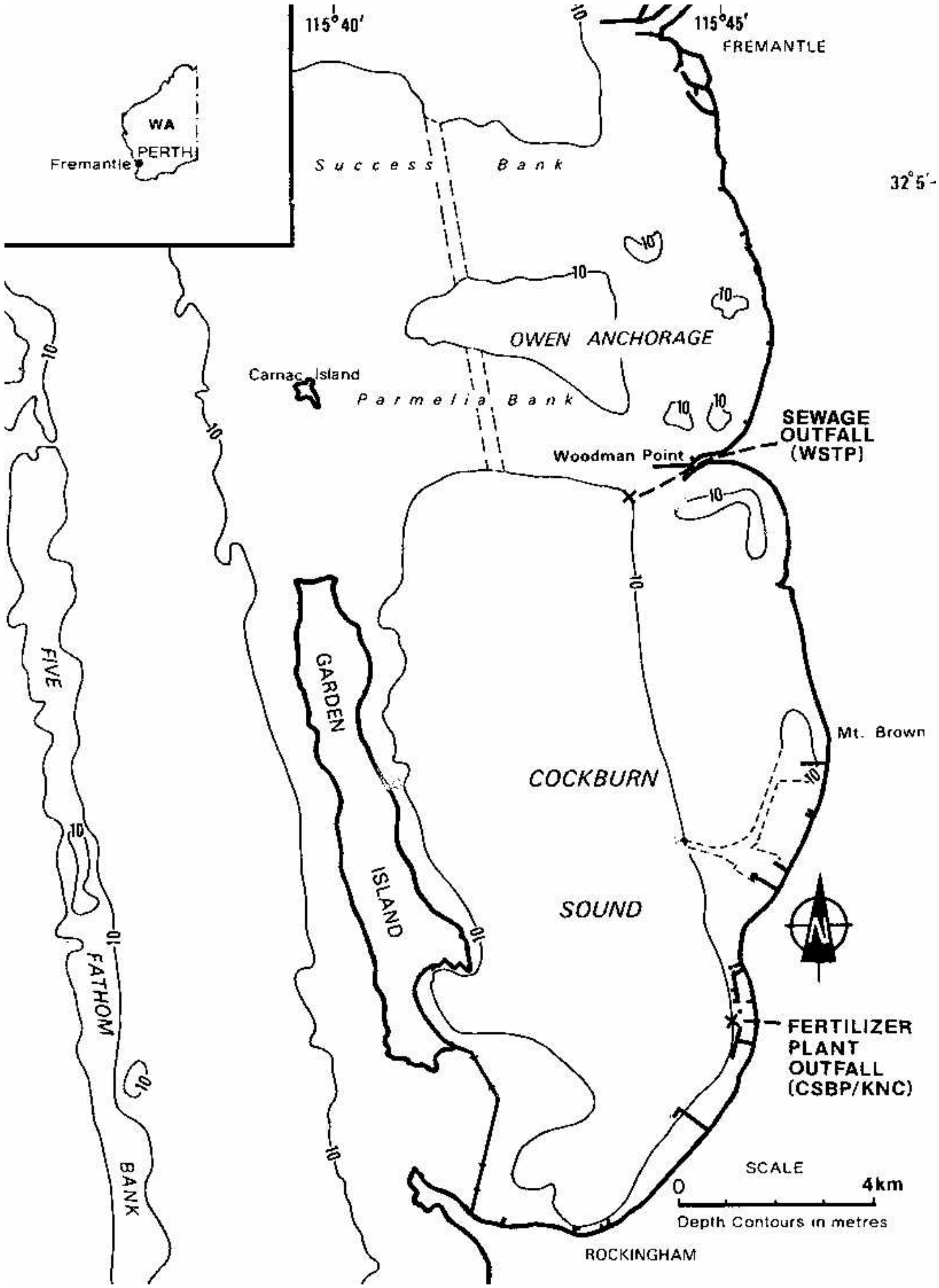


Figure 2.28 Map of Cockburn Sound, Western Australia.

(Southern Flats) between Garden Island and the mainland, which extends into the southern part of the deep basin of the Sound.

Up until the 1970s, the margins of the Sound and the Southern Flats were covered in dense beds of the seagrass genus *Posidonia*. These beds extended from the immediate subtidal to a depth of 8-10m. It has been calculated that from 1957 through to 1968 a gradual thinning of the seagrass beds took place and by 1972, all but about 7 km^2 of an original 40 km^2 had disappeared (Cambridge 1975). A reassessment in 1994 derived an estimate of 7.5 km^2, although it is known that there has also been a reduction in seagrass areas on the Southern Flat as well as Parmelia Bank to the north (DEP 1996).

Loss of seagrass was considered a result of diminished light, initially due to increased epiphytic loads on the seagrass, and then increasing phytoplankton concentrations. Both of these changes in primary production resulted from increased nutrient loads to the Sound from industrial sources.

Cockburn Sound has been extensively developed as the Outer Harbour of the port of Fremantle. Since 1954, much of the eastern shore has been developed for industrial use with adjacent port facilities. Some of the larger industries use water from Cockburn Sound for cooling purposes and a number of these industries also discharge wastes to the Sound. The mainland shore of Owen Anchorage is also heavily industrialised. As a noxious industries zone it has a number of abattoirs, food processing plants and tanneries. Some of their discharge wastes go directly to the Anchorage or to the groundwater (Murphy 1979).

Water Balance

In the present analysis, water exchange is based on numerical modelling of circulation, rather than on a water and salt budget. Exchange between the open ocean and Cockburn Sound may take place either in the south, through the two openings in the causeway which links Garden Island to the mainland, or through the northern opening of the Sound, across Challenger Passage (between Garden Island and Carnac Island) and Parmelia Bank (between Carnac Island and Woodman Point) (Figure 2.28).

At the time of the initial studies, exchange through the two southern openings was considered to be of minor importance (Steedman and Craig 1983) based on both numerical model predictions and field observations. Subsequent studies indicate that southern exchange is greater than originally anticipated (20-30% of total exchange – Mills and D'Adamo 1995). This needs to be borne in mind when considering the budgets presented, as they use modelled exchange rates based on the major exchange between Cockburn Sound and the open ocean taking place across the northern boundary.

The numerical model of Steedman and Craig was initially used to calculate northern exchange rates. The algorithm distinguished between water exchange by a small gyre on the western side of the Sound, and that of a larger gyre on the eastern side. As calculation of the total volume exchange between each cruise would have involved the expense of running the model for long periods, an empirical relationship was sought between predicted northern exchange and wind stress. Wind is the primary forcing function for water movement in Cockburn Sound (Steedman and Craig 1979, 1983).

A direct relationship between wind speed and direction and exchange volumes, predicted by the numerical model, was established by Steedman and Craig for the eastern gyre. Calculated exchange rates using only wind speed and direction correlated well with rates obtained from the numerical model. Predicted exchange associated with the western gyre was found to be reasonably consistent under most conditions, the average volume exchanged being 70×10^6 m^3 d^{-1} . Flux by this western gyre was therefore treated as a constant in subsequent calculation of northern volume flux.

Salinity differences between Cockburn Sound and the ocean are great enough to allow LOICZ-style mass balance estimates to be made. This needs to be done to ascertain the sensitivity of the model predictions to differences in exchange calculations.

Basis for Budget Calculations

The budgets presented here are based on 12 months of data collected in 1980 from 14 cruises over a period of 365 days. These data are part of a more extensive record of cruises undertaken between 1977 and 1983 (Chiffings 1987).

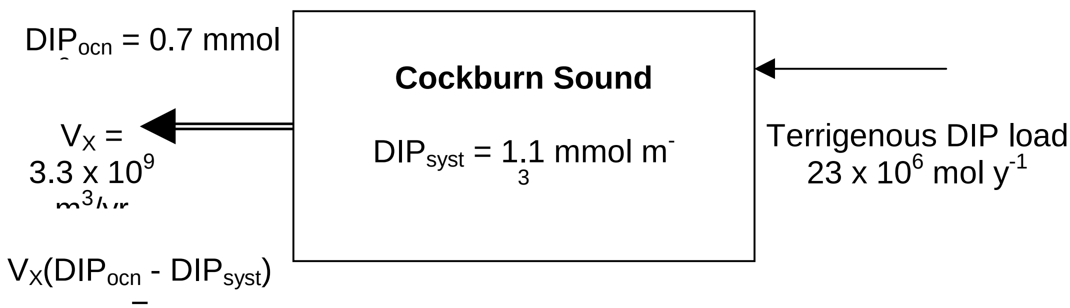
Input parameters to the material flux model used were the external nutrient load to the Sound, and the northern exchange volume flux. The external nutrient load for each monthly period was estimated by adding the loads from the fertiliser plant outfall and the sewage plant, as these had been identified as contributing some 72% of the DIN to the system and 89% of the DIP to the system. An estimate for other, previously identified sources of nutrients was added as a constant.

The mean exchange time for the 12 month period was 152 days. This is equivalent to an exchange volume (V_X) of about 9×10^6 m^3 d^{-1}, or 3.3×10^9 m^3 y^{-1}. Summer residence times were calculated as an average of 100 days ($V_X = 14 \times 10^6$ m^3 d^{-1}), compared with a winter average of 200 days (7×10^6 m^3 d^{-1}). This increase in oceanic exchange over the summer period, November to March, reflects strong sea breezes that lead to large exchange volumes through the northern opening. Large volumes were also exchanged during periods of frequent and persistent storms over the winter period (June-September). However, these storm events were less predictable and of short duration, with the passage of low pressure weather systems resulting in strong winds being interspersed with periods of calm conditions and therefore minimal exchange.

There are no natural drainage systems discharging to Owen Anchorage, Cockburn Sound or Warnbro Sound. Surface runoff is restricted to a small number of stormwater drains associated with urban and industrial development. Urban runoff contains nitrogen and phosphorus (Anonymous 1973), but in view of the magnitude of inputs from other sources, the possible contribution from stormwater drainage was assumed to be insignificant. The nearest river discharge is from the Swan Canning system, which discharges some 3.5 km north of Success Bank, the northern boundary of Owen Anchorage. During the budget period under consideration here, freshwater inflow from this river system was considered not to be a significant source of nutrients to the Sound. Since freshwater load is small, residual flow (V_R in the standard LOICZ calculations) can be considered to be zero, although this is not always the case (DEP 1996).

Groundwater flow in the shallow, unconfined aquifers to the south of Fremantle was estimated by flow-net analysis (Davidson 1981). These estimates, in conjunction with nutrient concentration data from bores close to the coast (Layton Groundwater Consultants 1979), have been used here to calculate inorganic nutrient loads to Cockburn Sound.

During the study, four industries were using water from the Sound for cooling purposes and waste disposal. Comprehensive studies were made of the composition of these discharges (Murphy 1979). An estimate for the Woodman Point Sewage Treatment Plant (WPSTP-primary treated sewage) is also included, based on weekly nutrient concentration and flow data provided by the Metropolitian Water Supply Sewage and Drainage Board. The sum of these loads to the system has been expressed as Y_{terrig} in the nutrient budgets.



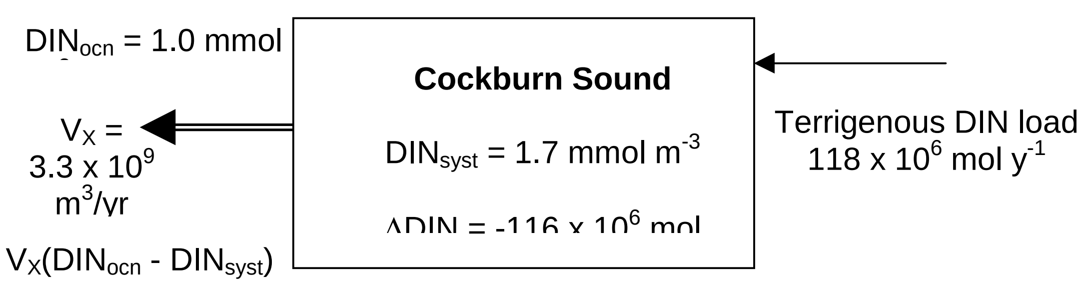


Figure 2.29 Annual DIP and DIN budgets during 1980, Cockburn Sound.

Nutrient Budgets
The original flux estimations were done for TN and TP and have been recalculated here for DIN and DIP using average ratios for the total external loads to the Sound (Figure 2.29). System concentration was calculated as the average of a sampling station located at the north-eastern end of the Sound. The coastal reference station was one located in the south-western part of Owen Anchorage.

Stoichiometric Analysis
Based on the above budgets and following the methods of Gordon *et al.* (1996), the following stoichiometric relationships were derived. Expressed on an areal basis, ΔDIP = -0.19 mol m^{-2} yr^{-1}, and ΔDIN = -1.1 mol m^{-2} yr^{-1}. The system appears to be a net sink as virtually none of the DIP and entering DIN is transported out of the system.

As most of the seagrass cover is now gone, the stoichiometric calculations are based on phytoplankton as the major DIP sink. Net ecosystem production (*p-r*) is thus calculated as -106 x ΔDIP = 20 mol m^{-2} yr^{-1} (or +55 mmol m^{-2} d^{-1}). This seems to be a high rate of net production, perhaps indicating an inorganic sink for the DIP or perhaps reflecting uncertainty in the water exchange.

Nitrogen fixation minus denitrification (*nfix-denit*) is estimated as the observed ΔDIN (ΔDIN_{obs} = -1.1 mol m^{-2} yr^{-1}) minus that expected based on ΔDIP. Again, based on the assumption that phytoplankton constitute the major primary producers in the system, we estimate ΔDIN_{exp} as 16 x ΔDIP; -3.0 mol m^{-2} yr^{-1}. This leads to an estimated rate of (*nfix-denit*) of +1.9 mol m^{-2} yr^{-1} (+5.2 mmol m^{-2} d^{-1}). This value seems a remarkably rapid rate of apparent net nitrogen fixation for this system, again possibly reflecting some uncertainty with the DIP budget.

Future Work
Cockburn Sound has been the subject of considerable study since 1980. Summer surveys of nutrient concentrations were undertaken in 1982/3, 1984/5, 1986/7 and then again in 1989/90 through to 1993/94, when another major study was undertaken (Anonymous 1996). During the latest intensive study a greatly refined quantitative understanding of the circulation and exchange processes was obtained (DEP 1996). In the intervening period between the studies, there has been a considerable change in both sources and total loads of nutrients to the system. In short, the Sound represents an opportunity to look more carefully at the long-term impacts of nutrient loadings on a temperate coastal ecosystem. New budgets will be developed as time permits, to fully explore these changes.

2.8 SWAN CANNING ESTUARY, WESTERN AUSTRALIA

Linda Kalnejais, Kathryn McMahon and Malcolm Robb

Study Area Description

The Swan Canning estuary (Site No. 29.; 115.90E, 32.00S) is located on the Swan Coastal Plain, Western Australia (Figure 2.30; see Figure 1.1). The total catchment of the estuary is 141 000 km^2, with the coastal plain covering 2 117 km^2. The majority of the catchment is cleared for agriculture, with intensive agriculture, grazing, light industry and housing on the coastal plain. The Swan Canning estuary runs through the city of Perth, which has a population of over 1.2 million people.

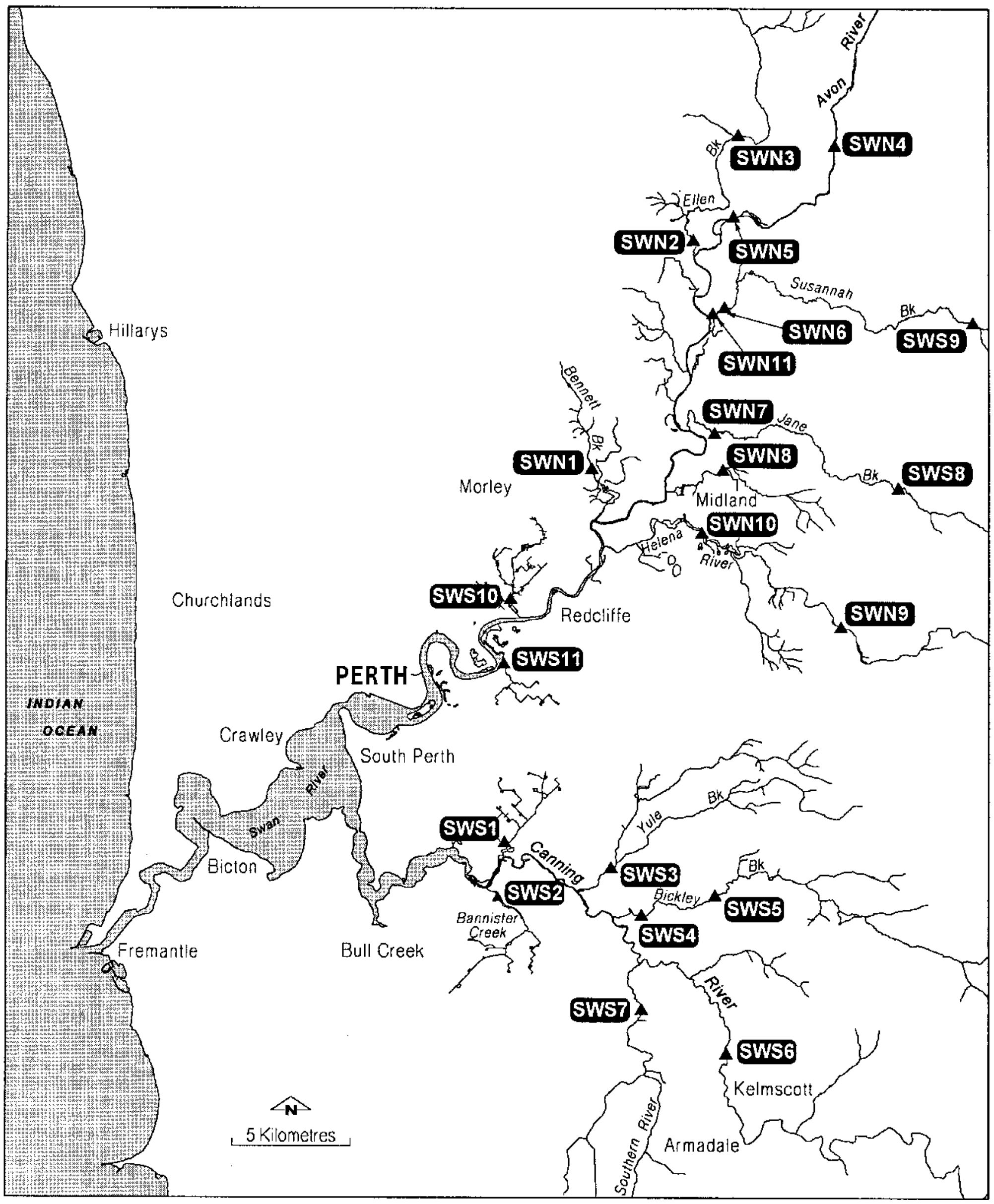


Figure 2.30 Swan Canning estuary. **(The sites shown are the Water and Rivers Commission water quality monitoring sites.)**

The Swan Canning estuary is permanently connected with the Indian Ocean at Fremantle. The estuary is deepest in the lower portions (10 to 20 m), with a shallow sill (3 m to 5 m deep) 5 kms from the harbour mouth which controls water exchange with the ocean. The estuary width ranges from 20 m to 1.5 km wide, with an average width of 230 m. From the Perth the river meanders upstream with an average depth of 2 m to 3 m, punctuated with holes of 5 m to 6 m. The tidal portion of the river is 50 km long, with a surface area of 33 km^2. The upper portion of the estuary occupies about 3 km^2 and has a mean depth of about 3.7 m, while the lower portion occupies 30 km^2 and has a mean depth of 4.9 m. Average tidal variation in the estuary is 0.5 m.

The Swan Canning estuary undergoes a distinct seasonal cycle of stratification (Stevens and Imberger 1996). Over summer (the low rainfall period), there is a gradual increase in salinity due to the upstream movement of the salt wedge. There is a strongly seasonal discharge of freshwater from the catchment, with most rain falling between May and September. Peak runoff occurs during winter and the saline estuarine water is usually flushed from the upper estuary, resulting in a vertically well-mixed water column in the shallow upper estuary. The freshwater discharge moves downstream as a buoyant plume and the lower estuary stratifies. As river flows reduce in October and November, the saline water migrates upstream as a salt wedge and salinity stratification is re-established in the upper estuary.

The dominant primary producers are phytoplankton which have been detected in bloom proportions in all sections of the estuary at various times, but are most significant in the upper reaches, blooming in spring, summer and autumn. Benthic macroalgae and the seagrass *Halophila* grow in the lower reaches of the estuary.

The Water and Rivers Commission has routinely monitored water quality in the Swan Canning estuary and its coastal plain catchment since 1994. Fifteen major catchment inflows have been monitored weekly for total and dissolved nutrients, when they were flowing. Nine estuary sites have been monitored for the above nutrients; vertical profiles measured for salinity, temperature, oxygen; and water samples collected to estimate phytoplankton density, on a weekly basis. The data collected from these programs has been used for LOICZ-style budgeting (Gordon *et al.* 1996), along with other data.

LOICZ Budgeting Guidelines

The following equation for the conservation of mass can be applied to any system:

$$\frac{dM}{dt} = \sum Inputs - \sum Sinks + \sum [Sources - Sinks] \qquad (1)$$

where M is the mass of a particular material. This equation represents the rates and quantities of material movement through the system and includes the effects of internal sources and sinks.

The LOICZ procedure applies this equation to water, salt and nutrients in order to obtain estimates on:

- the rate at which water moves through the system
- the rate at which carbon, nitrogen and phosphorus move with the water, and
- the importance of biogeochemical processes within the system.

The LOICZ procedure was applied to the entire basin of the Swan Canning estuary. The results and methods are discussed below.

Two-Layer Two-Basin System

The Swan Canning estuary behaves as a salt wedge estuary for much of the year. A two-layer LOICZ budget is thus used to model the system when stratified. To gain further information on the estuary it has been broken up into two basins: the upper basin and the lower basin, with the boundary at the Perth Causeway.

Water Balance

The LOICZ procedure assumes that the mass of water within the system stays constant over time:

$$\frac{dM}{dt} = \frac{dV}{dt} = 0 \qquad (2)$$

where V is the volume of the system. There are no internal inputs or outputs of water, so that equation (1) simplifies to

$$0 = \sum inputs - \sum outputs \qquad (3)$$

The generalised diagram used to represent the Swan Canning estuary is shown in Figure 2.31. Considering all the inputs and outputs to each box in the model, the water balance for the *n*th basin (where n=1 is the lower estuary and n=2 is the upper estuary) is given by the following:

Basin *n* surface layer

$$V_{Q^*\,n} + V_{G\,n} + V_{ent\ n} + V_{surf\ n+1} = V_{surf\ n} \qquad (4)$$

where

$$V_{Q^*\,n} = V_Q + V_P - V_E \qquad (5)$$

Basin *n* bottom layer

$$V_{deep\ (n-1)} - V_{deep\ n} = V_{ent\ n} \qquad (6)$$

Where the subscripts represent:

Q - river inflow
P - precipitation
E - evaporation
G - groundwater

$V_{ent\ n}$ represents the volume of water vertically entrained within the nth basin from the bottom layer to the surface layer; $V_{surf\ n}$ represents the volume of horizontal outflow from the nth surface layer and $V_{deep\ (n-1)}$ is the volume of horizontal inflow to the nth bottom layer from the (n-1)th bottom layer. $V_{deep\ (n-1)}$ when n=1 represents the volume of inflow from the ocean, ie V_{ocean}. For the final basin (n=2) $V_{deep\ n} = V_{surf\ n+1} = 0$.

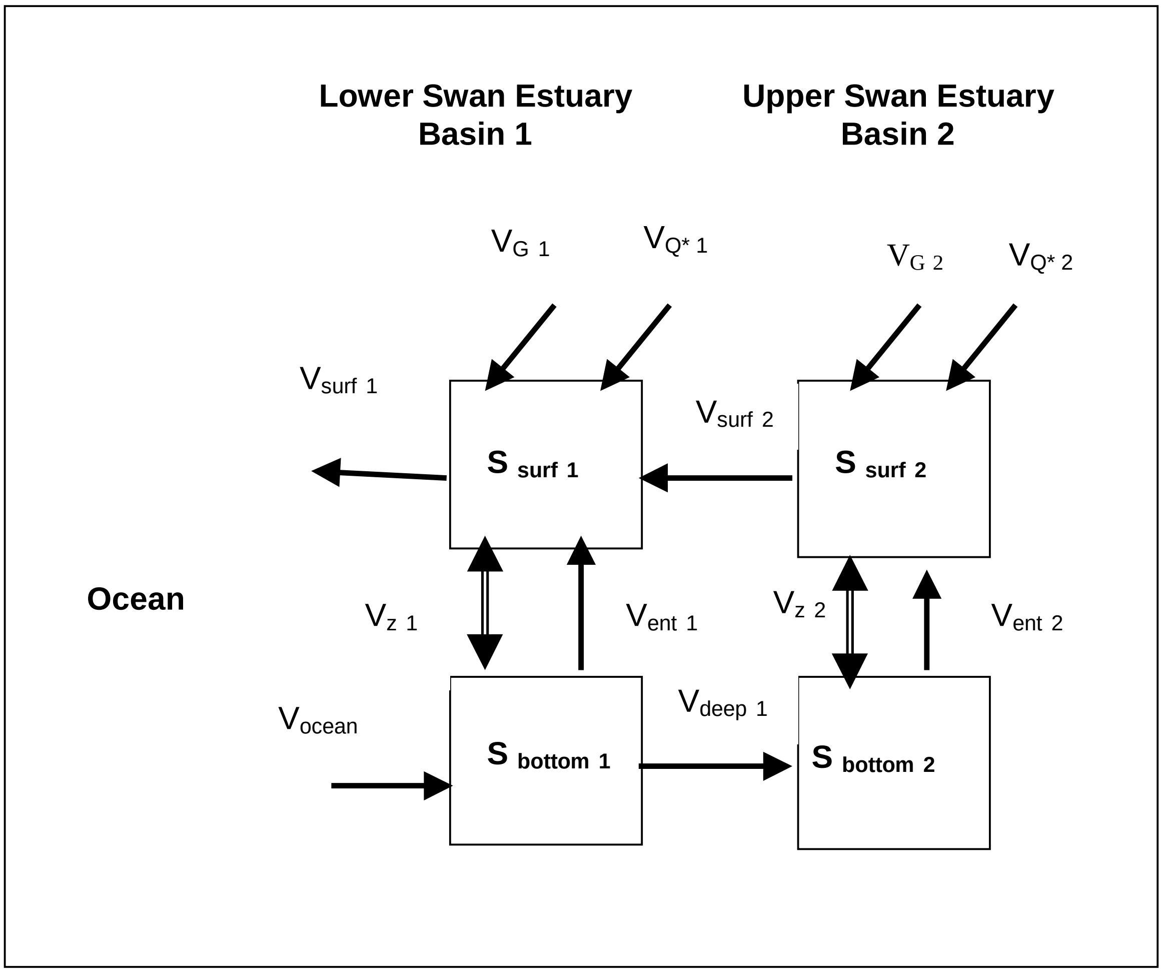


Figure 2.31 Generalised LOICZ model for the Swan Canning estuary.

<u>Salt Balance</u>

Salt is a conservative material, so equation (1) simplifies to equation (3), if it is assumed that the mass of salt in the system remains constant over the time-frame of interest. For a steady state salt balance for basin n, the salt exiting the surface layer equals the salt input from the bottom layer. That is:

$$S_{surf\ n}\, V_{surf\ n} = S_{bottom\ (n-1)}\, V_{bottom\ (n-1)}. \qquad (7)$$

Solving equations (4), (6) and (7) simultaneously gives the values for $V_{surf\ n,}$ $V_{ent\ n}$ and $V_{deep\ n}$. To complete the salt balance an additional vertical mixing term $V_{z\,n}$ is required to exchange surface and bottom water within a basin. $V_{z\,n}$ is thus given by

$$V_{zn} = \frac{V_{deep\,(n-1)}(S_{bottom\ n-1} - S_{bottom\ n})}{(S_{bottom\ n} - S_{surface\ n})} \qquad (8)$$

When a basin is well mixed the V_{zn} term tends to infinity and the two layer model reduces to a one layer model.

<u>Nutrient Budgets</u>

The conservation of mass equation, equation (1), can be applied to any material. For materials which undergo net transformations within the system, the Σ[sources-sinks] term is non-zero and represents the "non-conservative behaviour" of the system. Nutrients such as nitrogen and phosphorus are expected to display non-conservative behaviour. For a material Y, equation (1) can be for the n^{th} basin as:

Surface Layer

$$\frac{dVY}{dt} = Y_{Q_n}V_{Qn} + Y_{Pn}V_{Pn} + Y_{Gn}V_{Gn} + V_{ent\,n}Y_{bottom n} + V_{zn}(Y_{bottom} - Y_{surface}) + V_{surf\,n+1}Y_{surface+1}$$
$$- V_{surf\,n}Y_{surface} - Y_{En}V_{En} + \Delta Y_{surfn} \qquad (9)$$

Bottom Layer

$$\frac{dVY}{dt} = -V_{ent\ n}Y_{bottom\ n} - V_{zn}(Y_{bottom\ n} - Y_{surface\ n}) + V_{deep\ n-1}Y_{bottom\ n-1}$$
$$- V_{deep\ n}Y_{bottom\ n} + \Delta Y_{bottom\ n} \qquad (10)$$

where, ΔY =Y[sources-sinks].

Application to Swan Canning Estuary

Due to the highly seasonal nature of the rainfall within the Swan Canning catchment, the LOICZ budget has been applied over distinct seasons. The seasons are summer (January-May), winter (June-August) and spring (September-December). Data for 1996 was used in the budgets, so that the results of the LOICZ budgets could be compared with the nutrient fluxes estimated by Fredericks *et al.* (1997). 1996 was a relatively wet year, with the total annual rainfall of 889 mm, exceeding the long-term average of 796 mm (Bureau of Meteorology data for Perth Airport).

Water Balance

Table 2.22 Volume data sources for the Swan Canning estuary.

Quantity	Source
V_{Q2}	1996 flow data for Walyunga site (SWN4) divided by 0.7. Walyunga contributes 70% of annual flow to the Swan Canning estuary (Frank Davies, pers. comm.).
V_{Q1}	Canning River inflow is approximately a factor of 15 smaller than inflows from the Swan River (Swan Canning Cleanup Programme Draft Action Plan) i.e., Flow data for Walyunga/15.
V_P	1996 rainfall data from Bureau of Meteorology, Perth Airport Station multiplied by surface area of estuary.
V_E	17-year monthly mean of pan evaporation from Bureau of Meteorology, WA converted to estuary evaporation by Perth pan to lake coefficient of 0.9 (Surface Hydrology Section, pers. comm.).
V_{G2}	Gnangara groundwater mound south and Cloverdale aquifer discharge to estuary. From Davidson (1995)
V_{G1}	Gnangara groundwater mound south and Jandakot groundwater mound discharge to estuary. From Davidson (1995).
V_{Inlet}	158 $x10^6 m^3$ (from Chari Pattiaratchi, pers. comm.).

Table 2.23 Average depth of surface layer (metres) in each season of 1996, Swan Canning estuary.

Season	Lower Estuary	Upper Estuary
Summer	Well mixed	1.5
Winter	2.5	5 (entire water column)
Spring	5	2

Salt Balance

For the Swan Canning estuary it can be assumed that $S_P \sim S_E \sim S_G \sim S_Q \sim 0$. S_Q was originally included in the calculation for salt balance but due to the low salinity of the inflowing water (1 ppt), it was insignificant and was removed from the equation. This leaves $S_{bottom\ n}$ and $S_{surf\ n}$ for both basins to assign. With each season, the depth of each layer changes. The average depth of the freshwater layer in the two basins was estimated from an overview of the vertical transect data. The average depths of each layer are shown in Table 2.23. The average salinities of each layer and each basin were then calculated.

Nutrient Balances

The DIN and DIP concentrations required to evaluate equations (9) and (10) were taken from 1996 sampling data and the additional sources shown in Table 2.24. The individual layer concentrations were calculated as the average concentrations within that layer.

Table 2.24 Average nutrient values used in the mass balance, Swan Canning estuary.

Quantity	DIN (mmol m^{-3})	DIP (mmol m^{-3})	Source
$Y_P \sim Y_E \sim 0$	0	0	
Y_G	143.0	3.0	Appleyard (1992).
Y_{ocean}	0.4	0.06	South Metropolitan Coastal Waters Study 1996 & Perth Coastal Waters Study, Water Quality Data (Buckee *et al.* 1994).
$Y_{Q\ summer}$	33.4	1.9	Average of weekly data from all gauging stations in the Swan Canning catchment, 1996 data.
$Y_{Q\ winter}$	70.0	1.9	
$Y_{Q\ spring}$	32.6	1.6	

Results

The water and salt budgets for each season are shown in Figures 2.34-2.36. The key differences between each season are obvious: evaporation and groundwater were the dominant input terms in the surface layer of the upper basin over summer. The marine water input dominated the well-mixed lower basin, with most of the marine water exported back out to the ocean rather than transported to the upper estuary. The groundwater inputs were negligible over winter, with high freshwater inputs dominating the water balance. The water column in the upper estuary was completely flushed with freshwater, so that the upper estuary has been represented by a single box. The spring system represents a transition between the other two seasonal states.

The water exchange time within the entire Swan Canning estuary (the average length of time water stays in the system) for each season can be calculated by:

$$\tau = \frac{V_{Inlet}}{(V_{ocean} + \sum_{n}(V_{Q*} + V_G))} \qquad (11)$$

Table 2.25 gives the exchange times of the estuary for each season of 1996. The upper estuary is estimated to contribute 7% to the total estuary volume of 158 million m^3.

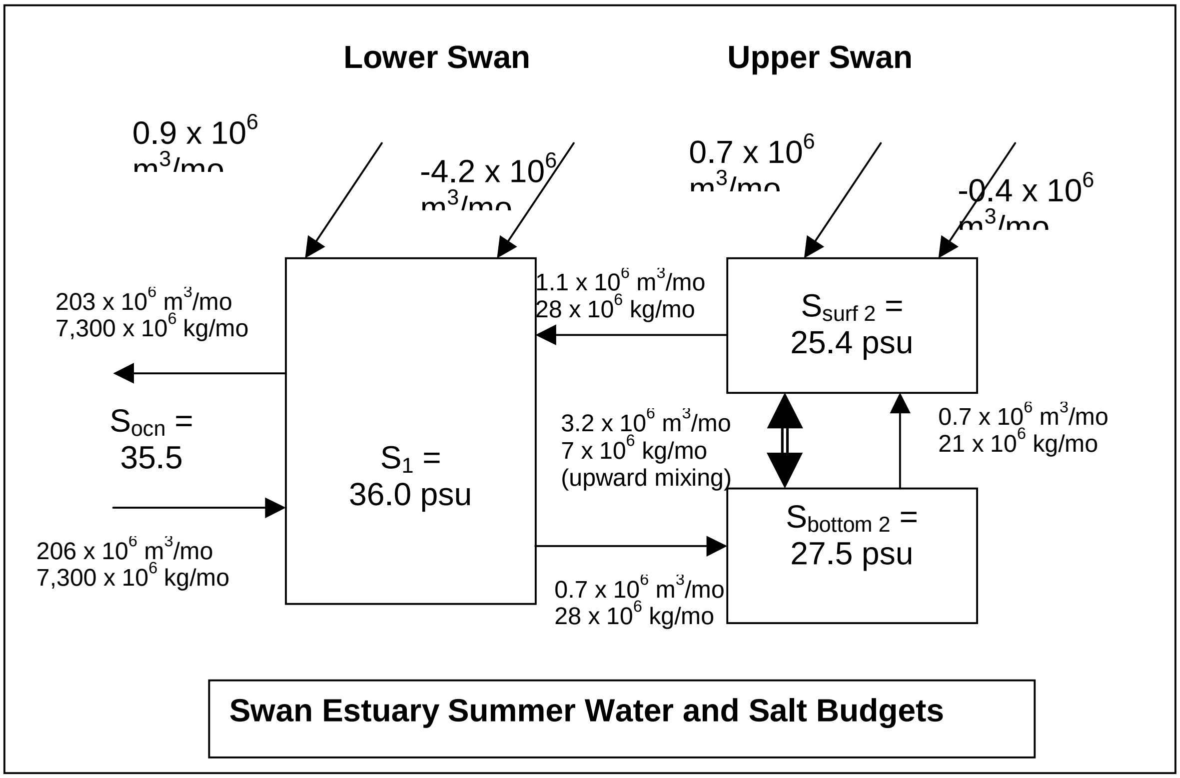


Figure 2.32 Summer water and salt budgets, Swan Canning estuary (see Figure 2.31 for details of arrows).

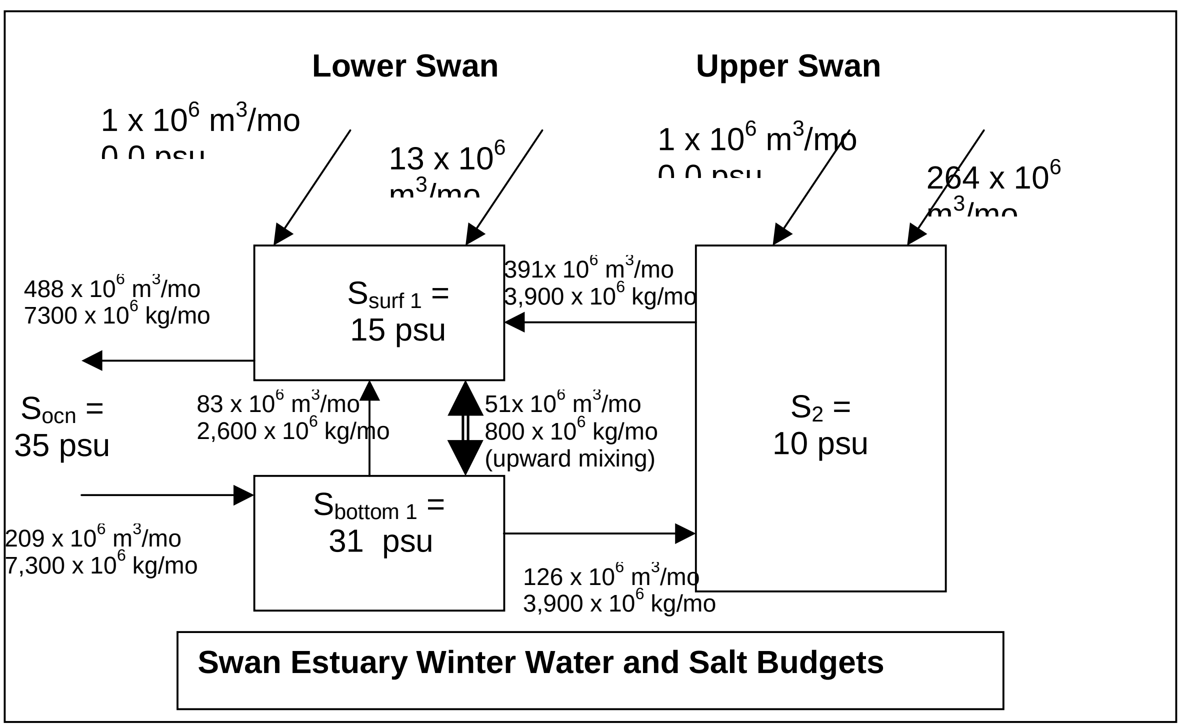


Figure 2.33 Winter water and salt budgets, Swan Canning estuary (see Figure 2.31 for details of arrows).

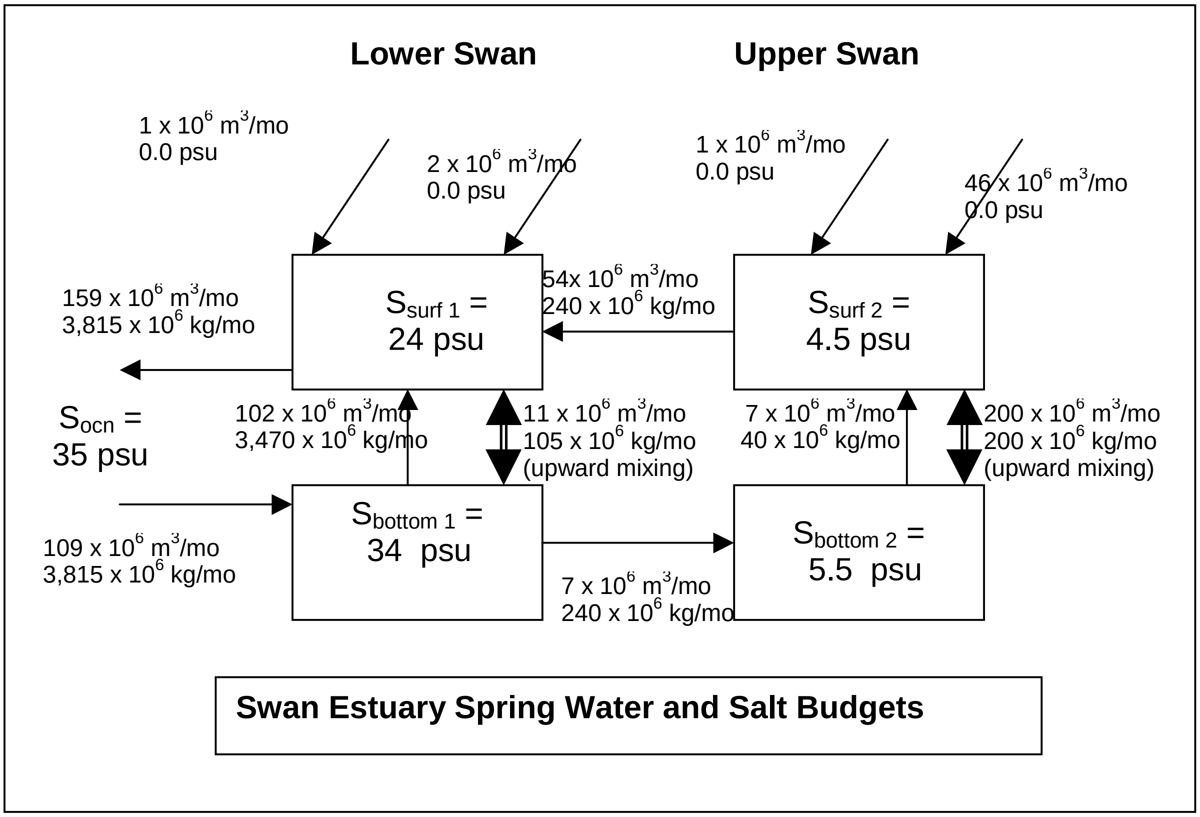


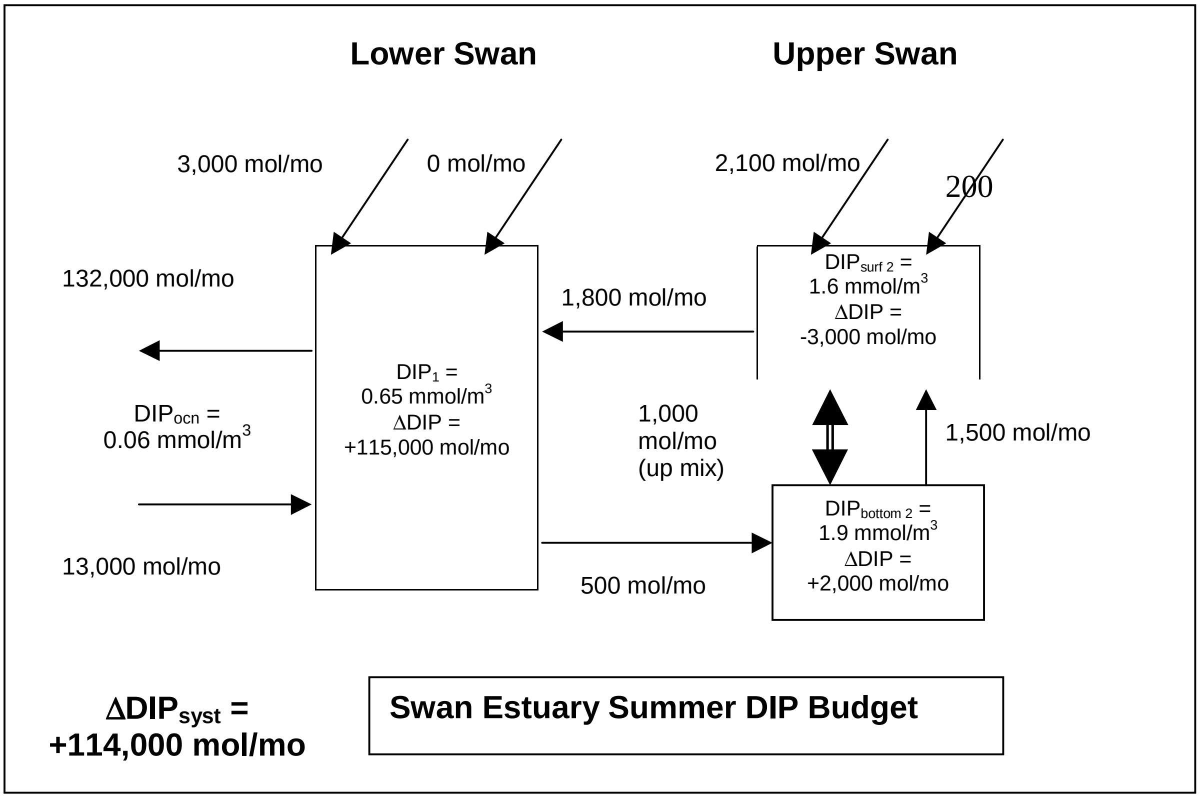
Figure 2.34 Spring water and salt budgets, Swan Canning estuary (see Figure 2.31 for details of arrows).

Table 2.25. Water exchange times for 1996, Swan Canning estuary.

<u>Season</u>	Exchange Time (days)		
Volume	**Lower Estuary** 147 million m^3	**Upper estuary** 11 million m^3	**Entire Estuary** 158 million m^3
Summer	22	235	23
Winter	7	1	10
Spring	13	6	30
Annual			**17**

<u>Nutrient Budgets</u>
The nutrient budgets for each season are shown in Figures 2.35-2.37. Due to the considerable seasonal differences in the water budgets there are corresponding differences in the nutrient budgets.

In summer the groundwater term dominated the external inputs of both DIN and DIP. The DIP budget indicates that in the upper basin much of the DIP was taken up in the surface layer (probably due to phytoplankton uptake or adsorption to particles and subsequent sedimentation). The bottom layer indicates regeneration of DIP that is then



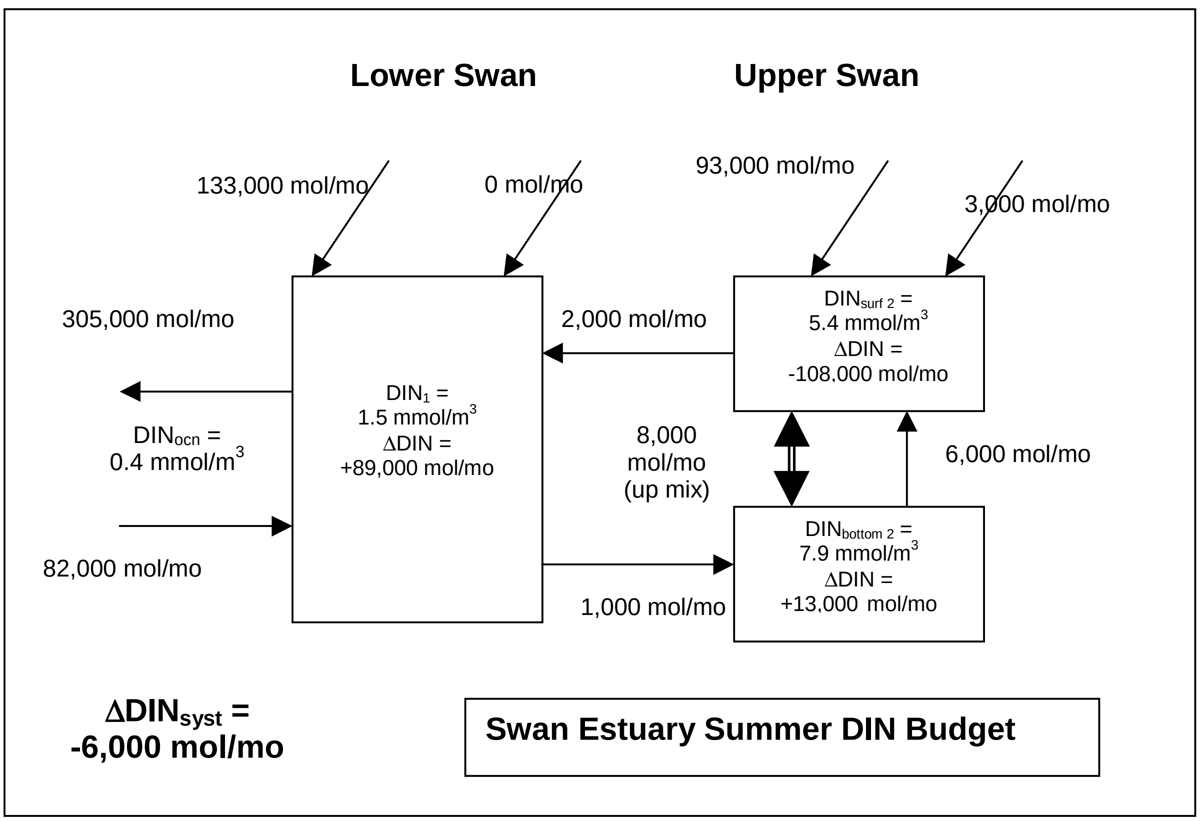
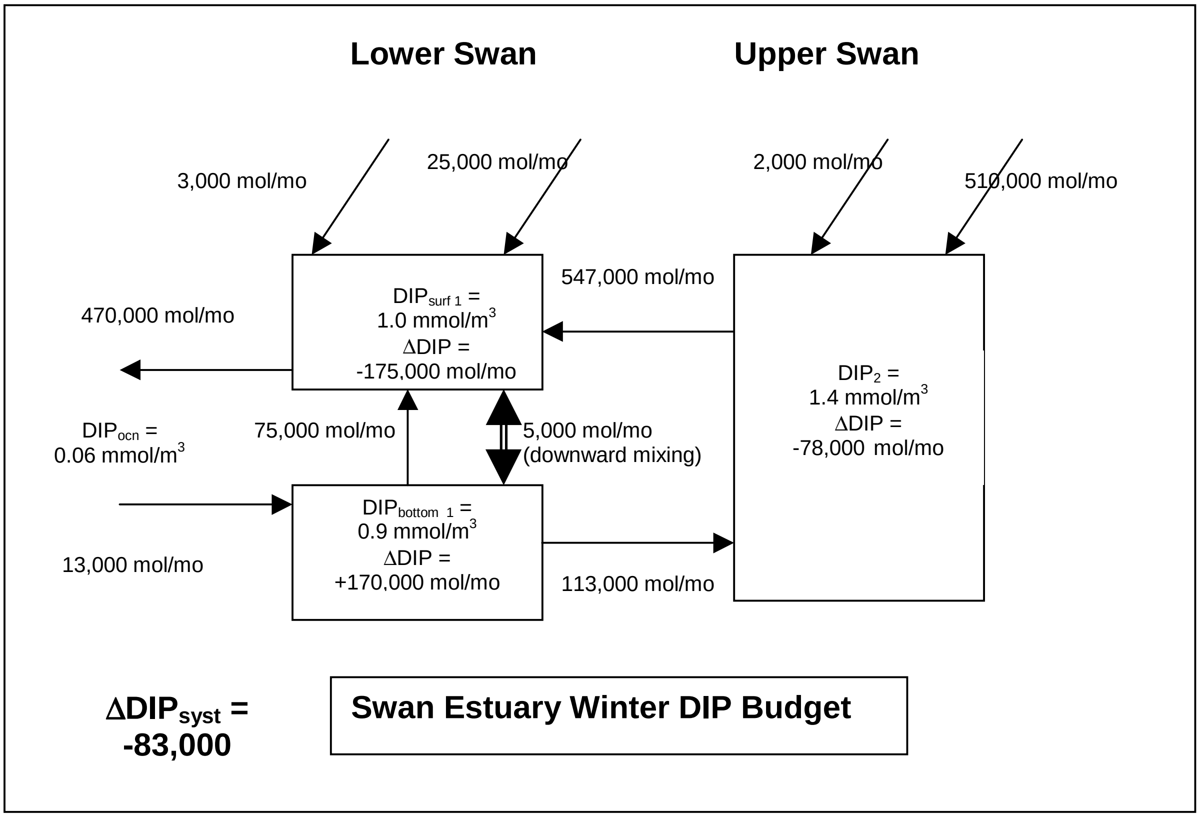


Figure 2.35 Summer DIP and DIN budgets, Swan Canning estuary (see Figure 2.31 for details of arrows).



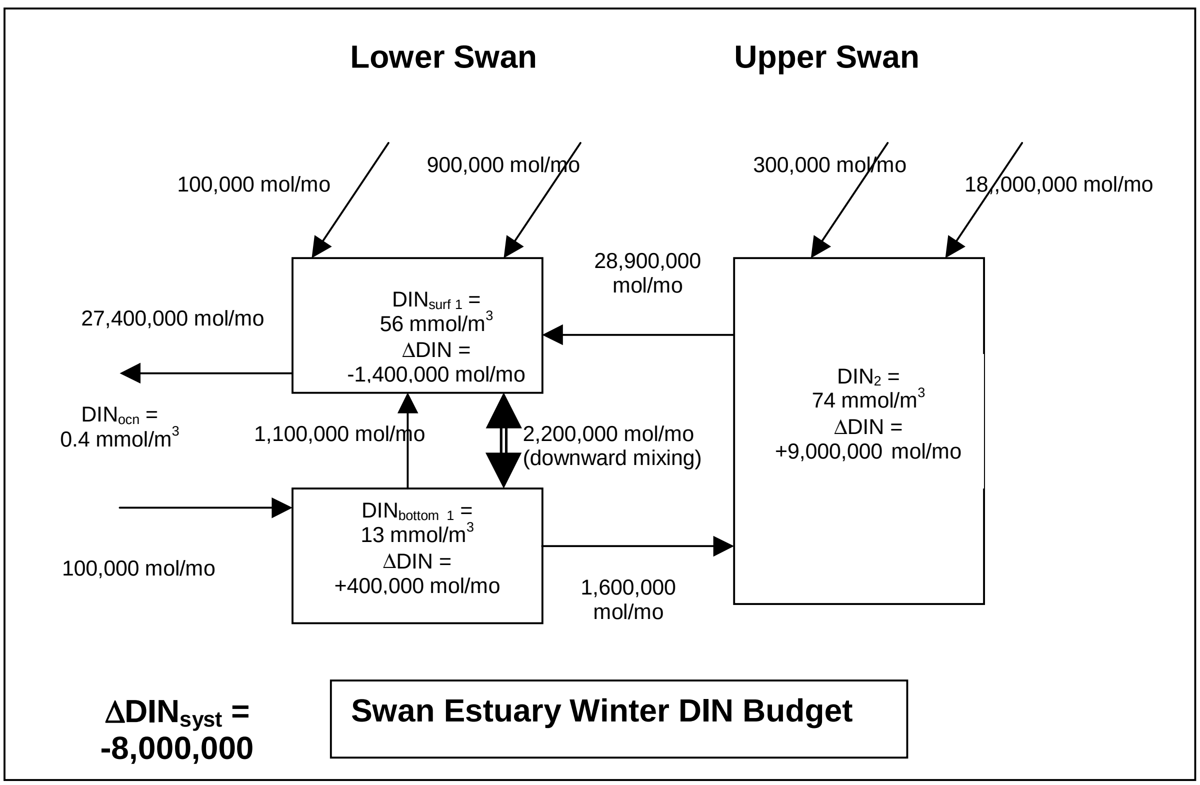


Figure 2.36 Winter DIP and DIN budgets, Swan Canning estuary (see Figure 2.31 for details of arrows).

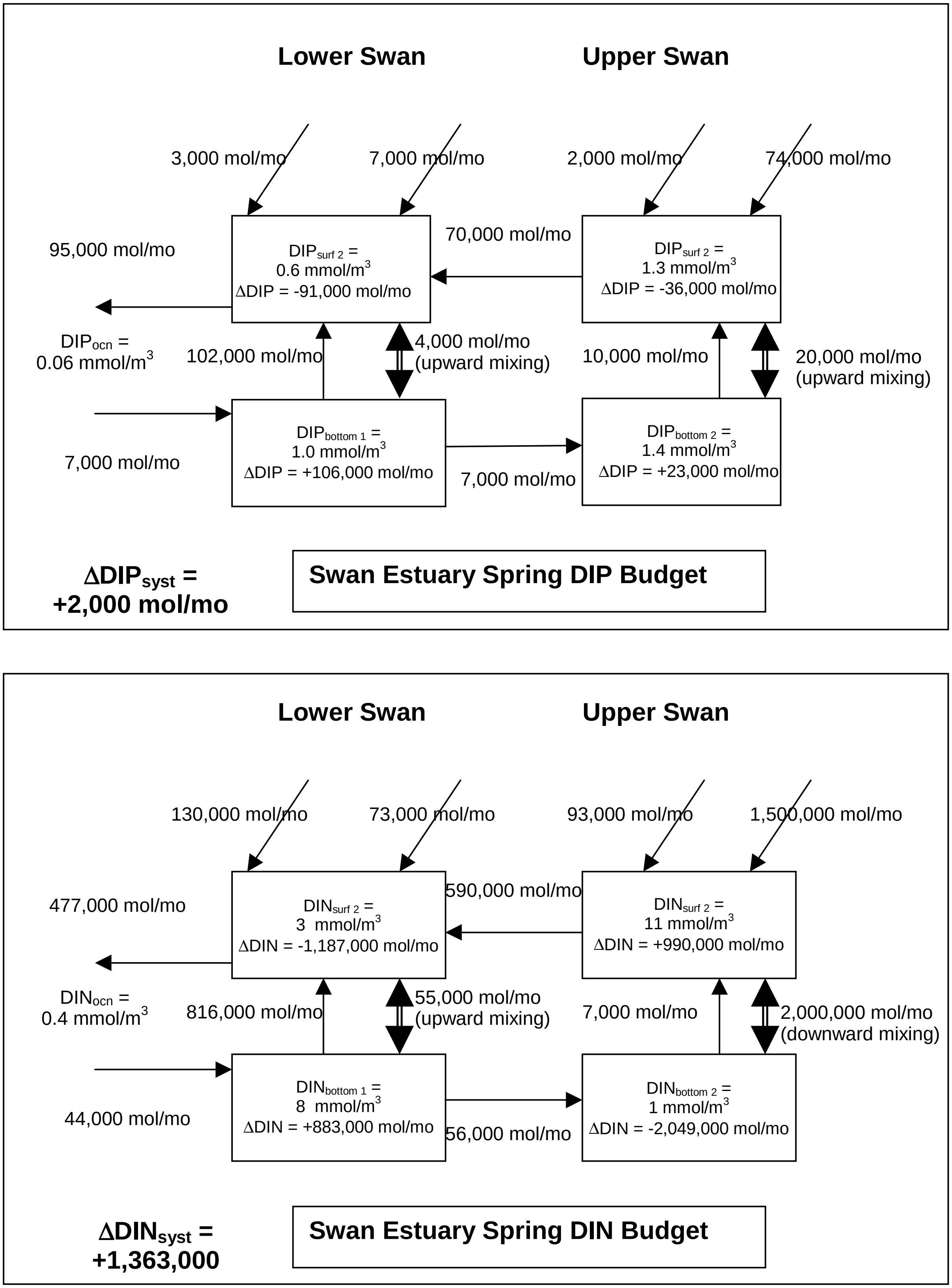


Figure 2.37 Spring DIP and DIN budgets, Swan Canning estuary (see Figure 2.31 for details of arrows).

transported into the surface layer. The upper basin overall was a small net sink of DIP. The DIP exported to the lower estuary from the upper basin was almost negligible compared to the large source of DIP generated within the lower estuary. Nutrient input from the ungauged urban catchment may contribute to this non-conservative source term. All of this DIP source term was exported from the estuary.

The summer DIN behaviour was similar to that of DIP. All of the groundwater DIN introduced into the upper estuary was retained within the surface layer of the upper basin. The bottom layer was a small source of DIN, which was exported to the upper layer. Again the lower estuary was a significant source of DIN, with the loadings from the source term and the groundwater inputs exported to the ocean.

The winter budgets have a much greater magnitude of nutrient fluxes, with large quantities of nutrients carried in with the freshwater inflows. The upper estuary acted as a sink for DIP and was a significant source of DIN. The surface waters of the lower estuary were also a net sink for DIP, while the bottom waters were a source of similar magnitude, so that the net non-conservative behaviour of P in the lower estuary was negligible. This probably represents net organic production in the surface layer followed by sedimentation and then P regeneration in the bottom sediments. The DIN species in the lower estuary showed a similar trend, with the surface waters acting as a sink and the bottom waters a source. For both species, export to the ocean was a significant mechanism for removal from the estuary.

The nutrient inputs from the catchment declined in spring, with reduced streamflow. There was uptake of DIP in the surface layer of both basins and release in the bottom layer. Most of the DIP in the upper estuary was exported in the surface layer to the lower estuary. More DIP was released in the bottom layer of the lower estuary than the total input from the catchment. Most of this DIP was entrained into the upper layer where it was either exported to the ocean or taken up within the surface layer. As in winter, the non-conservative behaviour of P in the lower estuary was balanced in the surface and bottom layers, so that overall the ΔDIP was negligible.

Nitrogen behaviour differed between winter and spring, with much of the nitrogen input from the catchment reaching the surface layer and subsequently mixed into the bottom layer of the upper estuary, where it was retained. The apparent production of DIN in the surface layer of the upper estuary may again be a result of nutrient input from the ungauged urban catchment. The surface layer of the lower estuary showed an uptake of DIN, while in the bottom layer there was a release of DIN. Export to the ocean was an important removal mechanism for DIP, but not for DIN, which was retained principally within the upper estuary.

A summary of the non-conservative terms in each season is given in Tables 2.26 and 2.27. The values are normalised by the area of each basin. The total area of the Swan Canning estuary is 33 km^2 with the lower estuary representing 91% of this area.

Table 2.26. Seasonal variation in non-conservative DIP fluxes (mmol m^{-2} d^{-1}), Swan Canning estuary.

Season	Lower Estuary			Upper Estuary		
	Surface	Bottom	Basin Total	Surface	Bottom	Basin Total
Summer	+0.13		+0.13	-0.03	+0.02	-0.01
Winter	-0.19	+0.19	+0.00	-0.93		-0.93
Spring	-0.10	+0.12	+0.02	-0.40	+0.26	-0.14
Annual Total (mmol m^{-2} yr^{-1})			**+17**			**-122**

Table 2.27. Seasonal variation in non-conservative DIN fluxes (mmol m^{-2} d^{-1}), Swan Canning estuary.

Season	Lower Estuary			Upper Estuary		
	Surface	Bottom	Basin	Surface	Bottom	Basin
Summer	+0.1		+0.1	-1	+0	-1
Winter	-1.6	+0.4	-1.2	+100		+100
Spring	-1.3	+1.0	-0.3	+11	-23	-12
Annual Total (mmol m^{-2} yr^{-1})			**-119**			**+10 461**

Stoichiometric Analysis

If it is assumed that DIP within the estuary undergoes negligible inorganic reactions and all the non-conservative behaviour is of biological origin, then the ΔDIP values in the Swan Canning estuary are a measure of the net production of organic matter in the system. The expected ΔDIN (ΔDIN_{exp}) would be ΔDIP multiplied by the N:P ratio of the reacting organic matter. If phytoplankton is assumed to be the principal form of organic matter then, based on the Redfield ratio, $\Delta DIN_{exp} = 16\ \Delta DIP$. Large differences between ΔDIN_{obs} and ΔDIN_{exp} are indicators of other processes that alter fixed nitrogen. As nitrogen fixation and denitrification are important processes in coastal systems, the differences are taken as a measure of net nitrogen fixation minus denitrification.

$$[N\textit{fix}\text{-}N\textit{denit}] = \Delta DIN_{obs} - \Delta DIN_{exp} \qquad (12)$$

The values of (*nfix-denit)* for each basin are shown in Table 2.28.

Table 2.28. Stoichiometric Analysis (*nfix-denit)* (mmol m^{-2} d^{-1}), Swan Canning estuary.

Season	Lower Estuary			Upper Estuary		
	Surface	Bottom	Basin	Surface	Bottom	Basin
Summer	-2.0		-2.0	-0.5	-0.3	-0.8
Winter	+1.4	-2.6	-1.2	+115		+115
Spring	+0.3	-0.9	-0.6	+17	-27	-10
Annual Total (mmol m^{-2} yr^{-1})			**-391**			**+12 413**

The stoichiometric analysis indicates that in summer the entire estuary was net denitrifying, with the greatest magnitude of denitrification occurring in the lower estuary. When the estuary was stratified in winter and spring, there was net denitrification in the bottom waters of each basin and a smaller amount of nitrogen fixation in the surface waters. Overall, under stratified conditions, each basin was net denitrifying. In winter when the upper estuary was fully flushed with river runoff and the water column was well mixed, a large net nitrogen fixation was indicated. In general, the lower estuary (which occupies most of the system and which has salinities near oceanic) appears to generate (*nfix-denit*) values which seem reasonable, while the upper estuary does not.

Net Ecosystem Metabolism

The net ecosystem metabolism (NEM = p-r) is calculated as the negative of ΔDIP multiplied by the C:P ratio of the reacting organic matter. Assuming the bulk of the reacting organic matter is phytoplankton, the C:P ratio is 106:1. Thus:

$$(p\text{-}r) = -106(\Delta \mathrm{DIP}) \qquad (13)$$

For the Swan Canning estuary, (p-r) is shown in Table 2.29.

Table 2.29. Net ecosystem metabolism (*p-r*) (mmol m^{-2} d^{-1}), Swan Canning estuary.

<table>
<tr><td>Season</td><td colspan="3">Lower Estuary</td><td colspan="3">Upper Estuary</td></tr>
<tr><td></td><td>Surface</td><td>Bottom</td><td>Basin</td><td>Surface</td><td>Bottom</td><td>Basin</td></tr>
<tr><td>Summer</td><td colspan="2">-14</td><td>-14</td><td>+3</td><td>-2</td><td>+1</td></tr>
<tr><td>Winter</td><td>+20</td><td>-20</td><td>0</td><td colspan="2">+99</td><td>99</td></tr>
<tr><td>Spring</td><td>+11</td><td>-13</td><td>-3</td><td>+42</td><td>-28</td><td>+14</td></tr>
<tr><td colspan="3">Annual Total (mmol m^{-2} yr^{-1})</td><td>-1 800</td><td></td><td></td><td>+12 900</td></tr>
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The lower estuary was apparently a net consumer of organic matter in summer. It is surprising that there was not a more significant production in the warm, light summer conditions. However this value for consumption may be inaccurate due to contributions of the ungauged drains to the positive ΔDIP term. In winter and spring the lower estuary was a net producer of organic matter in the surface waters throughout each season. The production was greatest in winter. The surface production was closely balanced by net consumption in the bottom waters, so that throughout the entire water column the (p-r) was close to zero.

In summer and spring, the upper estuary showed net production in the surface layer and net respiration in the bottom layer. The production in the surface layer exceeded respiration in the bottom layer, so that the upper estuary was a net producer of organic material. The production was highest in spring. The NEM in winter was the highest reported and indicated a production of organic matter. A higher production in winter compared with spring seems unlikely with the high turbidity and low temperature and light conditions in winter. It is likely that the NEM of the upper estuary in winter is incorrectly elevated by contributions to the ΔDIP term from inorganic and particle reactions and subsequent sedimentation.

General Discussion of Nonconservative Fluxes
Because of relatively short residence times in the upper estuary during winter and spring (Table 2.25), non-conservative flux calculations and stoichiometric balances are questionable. We consider the calculations based on the whole estuary, particularly those based on the annual balance, to be the most reliable.

Based on the total system mass balance (Figures 2.35-2.37), the total estuary ΔDIP is about +4 mmol m^{-2} yr^{-1}. This represents a very small portion of the terrigenous DIP input (about 77 mmol m^{-2} yr^{-1}), and is effectively indistinguishable from zero. Uptake of DIP in the upper estuary during the summer is slightly more than offset by DIP released in the lower estuary during the summer, so that when integrated over an annual cycle the system is nearly conservative with respect to DIP. Also based on those figures, the total estuary ΔDIN is about 1 130 mmol m^{-2} yr^{-1}. This represents DIN production equal to about 40% of the terrigenous DIN loading and is sufficient to suggest that there is a net DIN source within the system. The estuary total (*nfix-denit*) is calculated to be about +1 070 mmol m^{-2} yr^{-1}, suggesting net nitrogen fixation over an annual cycle. The calculated estuary total (*p-r*) is about -400 mmol m^{-2} yr^{-1}; that is, the system appears to be marginally net heterotrophic.

Comparisons with other work
The values derived in this LOICZ budget can be qualitatively compared to values experimentally measured by Fredericks *et al.* (1997) in a 1996 flux study of the Swan Canning estuary (see Appendix IVB). The numerical values cannot be directly compared, as the 1996 flux study provided instantaneous surface values, whereas this LOICZ budget has been averaged over an entire season with two depth layers. However, a qualitative comparison is valuable in assessing the LOICZ budget.

The summer survey of Fredericks *et al.* (1997) found a mid-estuary internal input of DIN. The surface layer of the upper estuary in the LOICZ model also indicates an input of DIN.

Fredericks *et al.* (1997) undertook two surveys over the time period used in the winter LOICZ budget. The survey in early winter was a low flow survey, while the second survey was a high flow survey. The nutrient behaviour under the two flow conditions was quite different and makes difficult comparison of the LOICZ winter budget. However, both winter surveys of Fredericks *et al.* (1997) showed that there was a net source of ammonium within the upper estuary. This agrees with the winter LOICZ budget that indicates a significant positive ΔDIN term.

The results of the 1996 flux study for the lower estuary indicated that under low flow conditions there was an uptake of DIN and a conservative behaviour under high flows. The LOICZ budget indicates a small uptake in the surface layer, but with most DIN exported to the ocean. The LOICZ result agrees well with the combined results of both surveys.

The LOICZ budget found a DIP uptake in the upper estuary, which does not agree with the 1996 flux study findings of conservative behaviour in both the low and high flow conditions. The LOICZ budget indicates that there was net removal of DIP in the surface waters of the lower estuary - under low flows there was input into the lower estuary and under high flows there was removal in the lowest section of the estuary and input in the middle reaches (part of the lower estuary LOICZ box). Due to the greater magnitude of the fluxes under the high flow situation it is likely that the high-flow scenario would have dominated the winter LOICZ budget. The two results are thus hard to compare due to the spatial variation in behaviour indicated by the flux study.

The spring survey of Fredericks *et al.* (1997) found an internal input of DIN (in particular ammonium) in the upper estuary and DIN removal in the lower basin. The surface boxes of the LOICZ model also indicate this to have been the case in spring. The flux study indicated that there was an internal input of DIP in the upper estuary and removal in the lower estuary. The LOICZ model differs in that it demonstrates a removal in the surface waters of both the lower and upper estuaries.

<u>Further Work</u>

This model could be improved by increasing the accuracy of some of the data. More accurate information could be obtained for groundwater volume inputs into the estuary and their concentrations of nutrients. Total volume of surface water entering the catchment could be improved by having a study undertaken on the contribution from the ungauged catchment and by updating the data from the gauged stations.