APPENDICES

APPENDIX 1 The response of Australian estuaries and coastal embayments to increased nutrient loadings and changes in hydrology.

Graham Harris

[This paper was presented as the opening plenary of the Workshop.]

Introduction

The Australian continent is old, weathered and largely dry, with the lowest runoff of any major continent (Williams 1975, Alexander 1985). Apart from the northern, tropical part of the continent where monsoonal rains produce high runoff, estuaries and coastal embayments in the southern, temperate and subtropical parts of the continent are generally marine environments with small freshwater inflows (e.g., Port Phillip Bay, Harris *et al.* 1996). The coasts of the south-eastern part of Australia are dotted with coastal embayments formed from drowned valleys during sea level rise since the last glacial period (Bayly 1975). Most of these embayments are cut off from the sea by sand bars and dune systems so that tidal exchange is also small and water residence times are long (Jennings and Bird 1967). The coastal sand bars frequently close off altogether during dry periods. Many Australian coastal systems therefore become slightly hypersaline during the summer period when evaporation is most rapid.

The oceans of the southern parts of Australia tend to be warm temperate or subtropical and poor in nutrients (Rochford 1975) so the marine inputs of nutrients are consequently low. Thus we have many generally marine ecosystems in coastal embayments which, because of the low freshwater inflows and tidal exchanges, tend, in their pristine states, to be oligotrophic and dominated by seagrasses. When freshwater inflows do occur they tend to be large and infrequent (the variance in rainfall in Australia is very high, Alexander 1985) and the water residence time, T_w, may intermittently go from hundreds of days to a few days or less. (It is recognised that the concept of a water residence time is sometimes difficult to apply to estuarine systems; Officer and Kester 1991, Oliveira and Baptista 1997). While these ecosystems evolved to cope with the rather extreme climate and climate variability of the continent they are nonetheless very sensitive to changes in T_w induced by extensive damming and regulation of the coastal rivers for water supplies and irrigation.

Because the water residence times of these systems are often long, the water quality is dominated by sediment-water column exchanges. The role of the benthos and microbial metabolism frequently dominates the ecosystem response to external loads – denitrification is very important in these systems (Harris *et al.* 1996). Because sediment nutrient pools and regrowth of seagrasses dominate the ecosystem response to intermittent loads, these systems are rarely at steady state and are always in a state of adjustment to the last extreme event which may have been years before.

Most of the Australian coastal embayments and estuaries are shallow – even Port Phillip Bay with an area of 2 000 km^2 has a maximum depth of barely 20 metres. This means that there is extensive contact between the sediments and the water column driven by

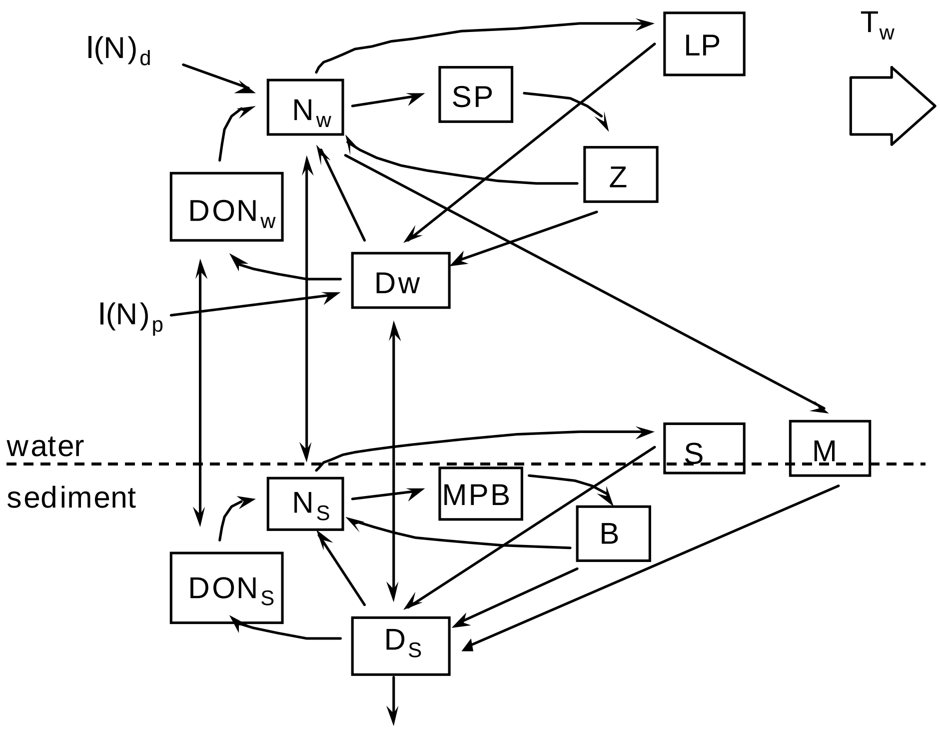

Figure I.1. A schematic diagram of the model. (The model was run with all stocks and flows calculated as nitrogen per unit area and for a 5m deep, fully-mixed water column. The exchanges between the sediment and the water column were calculated from the concentration difference and a diffusion coefficient. Nutrient (nitrogen) loads were split between dissolved [l(N)p] and particulate [l(N)d] fractions. Components as follows:
N_W, dissolved inorganic nitrogen in the water column; **DON_W**, dissolved organic nitrogen in the water column; **D_W**, detritus in the water column; **SP**, small phytoplankton; **LP**, large phytoplankton; **Z**, zooplankton; **N_S**, dissolved inorganic nitrogen in the sediment pore waters; **DON_S**, dissolved organic nitrogen in sediment pore waters; **D_S**, detritus in the sediment; **MPB**, microphytobenthos; **B**, benthic grazers; **M**, macrophytic algae (sea weeds); **S**, seagrasses.
Note that while M source nutrients from the water column, S draw nutrients from the sediments. The model calculated all light extinctions (m^{-1}) from sum of the biomass of phytoplankton (SP+LP), DON_W and D_W in the water column. Physiological parameters and grazing coefficients were derived from standard literature values ; Murray and Parslow 1997.)

wave action and wind driven currents. Because many systems are only a few metres deep, light intensities on the sediment surface are high and there is an extensive littoral zone. Macrophytic marine algae (seaweeds), seagrasses and microphytobenthos, in particular (Light and Beardall 1998), are important functional groups in these ecosystems. Where they are abundant, large beds of macrophytes may change the response of coastal ecosystems to nutrient loads so that the expected planktonic algal blooms are suppressed (Borum and Sand-Jensen 1996). These features are echoed in the model structure and its functional response (Figure I.1).

As might be expected, these coastal systems are very popular tourist destinations and some are beginning to suffer from the usual problems associated with agricultural and urban development – eutrophication, algal blooms, loss of seagrasses and overfishing. Luckily, because of low freshwater inflows and low population densities, many of the Australian coastal ecosystems are still relatively unpolluted so we have many pristine systems. Nonetheless, there are a number of coastal ecosystems that are beginning to shows signs of stress and some are seriously impacted (Scanes *et al.* 1997, Harris *et al.* 1998).

Predictive models

In an effort to understand some of the important interactions in these ecosystems and to begin to tease out some of the controls on ecosystem structure and processes, I built a simple ecosystem model for these systems (Figure I.1). Much of the conceptual development of this model was carried out during the Port Phillip Bay Study so I owe a debt of gratitude to the other members of the Study team (see Harris *et al.* 1996, Murray and Parslow 1997). Further details of the conceptual development of the model used in this paper may be found in Harris (1997, 1998). This model describes the ecosystems of the coastal embayments of southeastern Australia. One major functional group not included in this model is the mangroves that dominate coastal ecosystems in protected sub-tropical and tropical estuaries. There is no reason why further versions of this model should not include this group.

In brief, the model employs well understood relationships for photosynthesis, nutrient uptake and growth, grazing and sinking and uses well established literature values for half-saturation constants, maximum growth rates, light saturation parameters, grazing efficiencies and sinking rates (Murray and Parslow 1997). The model was written as a nitrogen model for the ecosystem because of widespread evidence of nitrogen limitation in Australian coastal waters, a fact that was thoroughly documented in the Port Phillip Bay Study (Harris *et al.* 1996). The model was calibrated and validated by comparisons with a number of well documented ecosystem responses to nutrient loads e.g., dissolved inorganic to total nutrient ratios, algal biomass responses, export production and the organic load to the sediment, responses of functional groups to external loads. All these empirical responses are described in the literature. Full details of this model will be published elsewhere.

The important objective of this work was to explore the ecosystem behaviour and responses to loadings and flushing through the interactions between the functional groups, rather than to develop new formulations for the responses of the groups themselves. An examination of Figure I.1 will show that broadly there are a set of pelagic interactions and a set of benthic interactions controlled by analogous sets of functional groups. This choice of model structure is outlined in Harris (1997, 1998). What emerges is a strongly non-linear response of the model to changes in nutrient loadings and hydraulic flushing which results from competition for light and nutrients between the organisms in the water column and those on the sediment surface. The model incorporates a simple empirical relationship between benthic denitrification efficiencies and the internal load of carbon and nitrogen from the water column (Harris *et al.* 1996, Murray and Parslow 1997). High denitrification efficiencies in the Bay were dependent on high bioturbation and irrigations rates by the macrobenthos.

Work in Port Phillip Bay confirmed what has been observed elsewhere (Smith *et al.* 1989, 1991, 1997; Smith and Hollibaugh 1989, 1997); that P:R ratios in estuaries and coastal waters can be estimating by examining C, N and P budgets and using Redfield proportions. Using a nutrient budgeting approach, Smith *et al.* (1989, 1991, 1997) have shown a close linkage between the net carbon balance of estuarine ecosystems and denitrification. Many estuaries are heterotrophic (Smith *et al.* 1997) but, from the data available, it would seem that temperate estuaries tend to be more autotrophic than tropical and subtropical systems. More Australian data are presented elsewhere in this paper which confirm this general pattern. In this paper, the model is used to examine how the various functional groups in the ecosystem interact with external loads and flushing times to recycle and retain nutrients within the ecosystem.

<u>Model results – ecosystem responses to nutrient loads and changes in flushing rates.</u>

1. Functional groups and ecosystem function.

It is possible to use the model to calculate retention coefficients (R_C%, as in Dillon and Rigler 1974), that is, the balance of nutrient inputs, system storage and export. The relationships between T_W and R_C% are well known for phosphorus and for lakes where it is a simple matter to measure the loads to and exports from confined water bodies (Dillon and Rigler 1974, Kirchner and Dillon 1976, Larsen and Mercier 1976, Ostrofsky 1978). Empirical work by Vollenweider and others showed that the best scalar for flushing rates is $1/(1 + \sqrt{T_W})$. The retention coefficients for nitrogen and for estuaries are less well known because nitrogen itself is less well studied, and tidal exchanges make it more difficult to calculate T_W for estuaries (Oliveira and Baptista 1997) – but see Johnson *et al.* (1995) for a mass balance nutrient budget for a coastal sea lough. Plotting R_C% versus $1/(1 + \sqrt{T_W})$ from the model (Figure I.2) gives results which look very like lakes – R_C% declines from about 80% at water residence times in excess of 100 days to between 25 and 50% at residence times of just a few days. R_C% declines at long residence times if the critical load is exceeded.

Figure I.2 The relationship between flushing time and retention of nitrogen. (At long flushing times, retention coefficients decline sharply if the critical load is exceeded and the ecosystem becomes plankton dominated (lower points, left of graph). Nitrogen loads of 8-10 mgN m^{-2} d^{-1} are sufficient to cause the "critical load" to be exceeded if the water residence time is long enough.)

Figure I.3 The relationship between flushing times and retention of nitrogen in the model ecosystem with either all functional group present or with macrophytes or benthic denitrification deleted. (Note that at short residence times the uptake of nutrients by macrophytes (macrophytic algae and seagrasses) dominates the retention of nutrients whereas at longer residence times benthic processes dominate. Note also that flushing dominates the retention of nutrients when denitrification is deleted and that the system exports nitrogen (negative retention) when flushed.)

Also using the model it is possible to calculate the effect of deleting certain functional groups. This gives an idea of the functional group playing the most important role in determining $R_C\%$ at various water residence times. The results from these simulations are displayed in Figure I.3.

At short water residence times (2-20 days), the macrophytes are the main functional group taking up and retaining the load whereas at longer residence times the benthic denitrification (resulting from sedimentation from the pelagic) is clearly the main determinant of the retention of nutrients in the system. This makes sense when it is remembered that short residence times will tend to flush out the plankton and swing the balance in favour of the littoral and demersal macrophytes.

2. Directional change and hysteresis in the response to nutrient loads.

The model quite clearly shows that the ecosystem has a highly non-linear response to nutrient load. There is, in effect, a "critical load" beyond which the system changes markedly and it is difficult to restore the oligotrophic functional groups and system function. This hysteresis is related to the finite capacity for denitrification in the system driven by organic loads to the sediment. Port Phillip Bay showed this response clearly (Harris *et al.* 1996).

The use of both small and large phytoplankton functional groups ensures that blooms of large phytoplankton result from increased nutrient loads (as is observed in Australian coastal waters, Hallegraeff 1981, Hallegraeff and Reid 1986) and that the organic load to the sediment rises sharply with increasing nutrient loads. These increased nitrogen loads to the sediment are initially efficiently denitrified. Once the organic loads to the sediment from pelagic production are sufficient to induce anoxia in the sediments both nitrification and denitrification cease and the sediment pools of nitrogen build up (Figure I.4). The decomposed nitrogen is then returned to the water column as an internal load. This further stimulates phytoplankton blooms resulting in the highly non-linear response of the system.

Figure I.4 The effects of rising and falling loads on both large phytoplankton (LP) and the dissolved inorganic nitrogen in the sediments (DIN). (The pool sizes rise by the lower route and fall via the upper points when the nutrient load is reduced after inducing eutrophic conditions.)

Lakes show a similar increase in internal phosphorus load from the sediment once eutrophication has been induced (Nurnberg 1984, 1988). The situation is exacerbated in shallow estuaries and embayments where there is good contact between sedimentary nutrient pools and the water column.
Blooms of phytoplankton in the water column lead to the shading off and death of the seagrasses (as has been observed in Western Australia by Silberstein *et al.* 1986). The sequence of events in Australian coastal embayments and estuaries, as nutrient loads are increased, is for the seagrasses to first show epiphytic overgrowths. This is followed by the development of macrophytic algal beds (which are often free-floating and which drift with the tide, Harris *et al.* 1996), and finally by the development of phytoplankton blooms.

When eutrophic, the system switches from a clear, macrophyte-dominated system with high nutrient retention and high denitrification efficiencies to a turbid, phytoplankton-dominated system with low denitrification efficiencies, high nutrient concentrations in the water column and higher exports (Figure I.2). Lakes are well known to exhibit similar behaviour (Blindow *et al.* 1993, 1997). The model results show the same switch between states for estuaries (Figure I.5). There are many parallels between shallow lakes and estuaries.

Figure I.5 The inverse relationship predicted by the model for total phytoplankton biomass and macrophyte biomass in the 5m water column. (The points are plotted as the biomass of each functional group after 500 days of simulated time with various external nutrient loads.)

Because of the cessation of benthic denitrification once the “critical load” has been reached (and the consequent build-up of sedimentary nutrient pools, Figure I.4), the ecosystem shows marked hysteresis to increasing and reducing nutrient loads (Harris *et al.* 1996, Murray and Parslow 1997, Parslow 1998). Until benthic denitrification has

Figure I.6 The relationship between nitrogen load and the response of the dominant functional groups. (Under slowly increasing loads the seagrasses are initially dominant followed in order by macrophytes and finally phytoplankton blooms (upper figure). This is consistent with observations in Australian coastal ecosystems. If the “critical load” is exceeded and the phytoplankton blooms are produced, then the load has to be reduced considerably to achieve the desired oligotrophic state (lower figure). This may not always be possible.)

Figure I.7 The change in system state with constant load and varying flushing time. (At the left [flushing time = 25 days] the water column is clear, nutrient levels are low, denitrification efficiencies are high, macrophytes are abundant and algal blooms are absent. At the right [flushing time = 50 days] algal blooms appear, the water column is turbid, denitrification efficiencies are low and the nitrogen concentration in the water column rises, macrophytes disappear.)

been re-established (and this requires a marked reduction in load to reduce the production and sedimentation of organic matter to the sediments, as was observed in Port Phillip Bay, Harris *et al.* 1996), the system stays in a eutrophic state. Here, the sedimentary nitrogen fluxes are dominated by ammonia until the external nutrient load has been markedly reduced. Once benthic denitrification ceases, the only way to eliminate the excess nutrient is by flushing – either by the tide or by low nutrient freshwater inflows. Degraded coastal embayments such as the Gippsland Lakes in Victoria (Harris *et al.* 1998) export ammonia to the ocean on the falling tide (Figure I.3).

The ecosystem response to nutrient loads therefore depends on the history of extreme events and the ecosystem state when the load is applied. Increasing and decreasing loads produce quite different responses by the functional groups (Figure I.6). Simple visual observations of which functional groups are present can be useful indicators of ecosystem state (Scanes *et al.* 1998) but depend on the marked hysteresis in response to changing loads. Certainly it is difficult to restore degraded coastal ecosystems - in cases such as Port Phillip Bay the reduction in nitrogen loading required to restore the ecosystem once the critical load is exceeded cannot be achieved without removal of the city of Melbourne from the catchment!

3. The importance of water residence times (and the lack of long term steady state).

One surprising outcome of the model was its extreme sensitivity to water residence times and to flushing. On reflection this is, perhaps, to be expected. Variability in T_W leads to changes in the competition between the pelagic and benthic groups for light and nutrients and alters the removal rates of the pelagic components (Figure I.1). At long T_W, the pelagic groups are favoured by giving phytoplankton populations time to build up and shade-off the benthos. At short T_W, the pelagic groups are flushed out clearing the water for the seagrasses. At intermediate nutrient loads, it is possible to switch the ecosystem state from turbid and plankton-dominated to clear and macrophyte-dominated merely by changing the water residence time from 25 to 50 days (Figure I.7).

This means that in coastal Australia the changes in T_W brought about by river regulation, water extraction, dams and diversions have probably had as much of an impact on the coastal ecosystems as has the increase in nutrient loads due to urban and agricultural development in coastal catchments. By changing the frequency distribution of the already infrequent freshwater inflows and by increasing the water residence times, we have increased the probability of planktonic algal blooms even before any changes in nutrient loads. Through deforestation and the growth of urban and agricultural land use, we have increased the turbidity of freshwater inflows and therefore increased the negative impacts on seagrasses and macrophytes in the shallow coastal embayments. By changing the hydrology and water quality of Australian rivers so markedly, we have altered the ecology of Australian coastal systems in ways that we do not presently fully understand.

Given the long time-scale of turnover of the sediment nutrient pools and the similarly long time required for the regrowth of seagrasses, then these coastal ecosystems are never at steady state, being always responding to the most recent extreme event in the catchment. Most of the south-eastern region of Australia is strongly impacted by alternating droughts and floods resulting from El Nino/Southern Oscillation events at time scales of seven to ten years. This climate variability has a strong impact on freshwater, estuarine and coastal marine ecosystems (Harris *et al.* 1988, Harris and Baxter 1996).

Conclusions

This simple exercise in estuarine modelling has demonstrated that it is possible to reproduce most of the observed systems dynamics through manipulating the interactions between the major functional groups. Interactions between functional groups yield the marked non-linearity of the ecosystems to nutrient loads. The changes in benthic denitrification ensure that a "critical load" is observed, beyond which it is difficult to restore the system. As well as reproducing the responses of the functional groups to changes in loads, this approach also adequately reproduces whole system properties such as nutrient retention coefficients. Nutrient budgeting approaches and the biogeochemistry of major elements, such as reported elsewhere in this volume, will depend on the ecosystem state in these coastal ecosystems. Models such as this can be used to explore system function as well as the response of coastal ecosystems to various management options.

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APPENDIX II Australasian coastal systems overview

B. N. Opdyke, S. V. Smith, B. Eyre, D. T. Heggie, G. G. Skyring, C.J. Crossland, and J. Zeldis

The contiguous Australian continent has 30 000 km of coastline (at a resolution of about10 km). It has been estimated to have over 750 estuaries and embayments along this margin. Australia spans 30 degrees of latitude and is unique among "first world" countries because of the substantial tropical and subtropical areas it occupies (Bucher and Saenger 1991). To expand this description to all of Australasia, we must add approximately 15 000 km of coastline for New Zealand and 5000 km of coastline for Papua New Guinea. Thus, the Australasian region has approximately 50 000 km of the world's ~600 000 km coastline. If we consider that much of the biogeochemical and biotic activity of the world's coastal zone is likely to be confined to a relatively narrow coastal strip (<< shelf width), then the coastline length becomes one measure of the relative importance of a particular coastal zone at a global scale.

Some 86% of the Australian population lives in coastal regions, with beaches and marine activities holding an important role in Australian culture and activities. Similarly, New Zealand has a heavy population weighting toward coastal regions, with nearly all its major cities bordering the sea. Papua New Guinea's population, by contrast, is not primarily on the coast, with 80% living in the more mountainous regions. The total human population of the Australasian region is about 25 million.

Australian fisheries, like those in many nations around the world, have been over-exploited and mitigation of population pressures on coastal ecosystems has only recently been tackled as a major issue for state and local governments. Moreover, it will be demonstrated in the discussion to follow that land-based anthropogenic activities have significant impact on Australasian coastal systems. Similarly, maritime activities are central in New Zealand society, and coastal fisheries, mariculture and recreational use of the coastal zone are important. Land use changes associated with urbanisation and agriculture have affected New Zealand estuarine systems.

If salinity, evaporation, rainfall and runoff data are examined to calculate the ratio of average estuarine salinity (S_{syst}) to the oceanic end-member salinity (S_{ocn}), this ratio (S_{syst}/S_{ocn}), can be used as an indicator of freshwater storage. The flushing times of the freshwater from an estuary can also be established by looking at the integrated inventories of freshwater in the estuaries ($[S_{ocn}-S_{syst})/S_{ocn}]$ x $V_{syst)}$, and the net precipitation + runoff-evaporation i.e., the net freshwater input rate. This, of course, is implicit in the LOICZ Biogeochemical Modelling Guidelines (Gordon *et al.* 1996) and is particularly useful in classifying Australian coastal systems.

Extending the classification of Eyre (1998) and using reasoning suggested by Heggie and Skyring (unpublished), we can subdivide the Australasian coastline into 8 major regions (Figure II.1).

Figure II.1 Classification of Australasian coastal zones.

I. Wet Tropical:- The high rainfall and runoff region in Papua New Guinea and smaller islands of Australasia north of the continent of Australia. Rainfall in Papua New Guinea typically is high all year, with annual rainfall exceeding 4000 mm $year^{-1}$ over much of the highland area. Most rivers of the region flow out of small, mountainous watersheds, and have some of the highest water and sediment yields of the world's rivers. These regions will typically show $S_{syst} << S_{ocn}$, with riverine influence extending onto the open shelf.

II. Wet/Dry Tropical:- Includes the monsoonal region of Northern Australia. Darwin is the only major city in the region. It is a sparsely population coastline, with extensive fisheries and potential for petroleum exploration. It is characterised by hot, wet summers and warm, dry winters. These regions therefore show $S_{syst} << S_{ocn}$ in the summer and $S_{syst} > S_{ocn}$ in the winter.

III. Wet/Dry Subtropical:- The coast of Queensland south of about 15°S, extending down into northern New South Wales falls into this category. The city of Brisbane is within this region. Much of the region is characterised by intensive coastal agriculture. This region is also monsoonal, with warm, wet summers and cool, dry winters. Generally, estuaries show $S_{syst} << S_{ocn}$ in the summer and $S_{syst} < S_{ocn}$ in the winter.

IV. Transitional:- The central south-eastern seaboard of Australia, extending north and south of Sydney, can be classified by this regime. It receives a more consistent rainfall than the arid tropical/subtropical regions, but river flow to bays and estuaries is low in the winter months. Typically, estuaries exhibit $S_{syst} < S_{ocn}$.

V. Wet Temperate:- A large proportion of Australians live on this coastal region (including inhabitants of Melbourne and Hobart). The region includes most of south-eastern Australia and Tasmania. It also typifies the North Island and the northern part of the South Island of New Zealand, and including its most populous city, Auckland. The estuaries typically yield a land-to-ocean gradient in salinity (S_{syst} << to < S_{ocn}), and may be considered freshwater dominated.

VI. Dry Temperate:- Includes the region of the Great Australian Bight, containing two large gulfs, numerous smaller embayments, and the South Australian city of Adelaide. Virtually no surface runoff and little groundwater reach the coast. Hence, estuaries show S_{syst} > to >> S_{ocn}.

VII. Mediterranean:- Contains much of the southern and western margin of Australia, and includes the city of Perth. These areas receive precipitation in the winter months and very little rainfall during the rest of the year. Estuaries show seasonal changes in salinity gradients with $S_{syst} < S_{ocn}$ during May through October, and $S_{syst} > S_{ocn}$ during the evaporative summer months.

VIII. Dry Tropical/Subtropical:- Typical of the northwest of Australia, and is extremely dry, with rare precipitation largely associated with monsoonal cyclones. Population density is very low, with no major cities. This area is typically dominated by $S_{syst} > S_{ocn}$ systems.

This estuarine classifications shows strong linkages with the climate zones defined for the Australian interior (Figure II.1). Moreover, there are important anthropogenic modifications to the pattern outlined. Because most of the Australian interior is relatively to very arid, and because much Australian land use involves extensive irrigation, natural flow of Australian river systems tends to have been greatly modified. Flow in some river systems is lowered, in some cases, dramatically below natural flow. This is particularly true during base flow conditions, although major floods can still deliver large water discharges. Secondly, flow in major river systems is strongly regulated.

It follows that an evaluation of coastal systems of Australia must include not only the climate linkages, but also consideration of land use in the river drainage basins and locations of major population centres.

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APPENDIX III Implications of spatial and temporal variation for LOICZ biogeochemical budgets

Ian T. Webster, John S. Parslow and S. V. Smith

Introduction

The LOICZ Guidelines for constructing biogeochemical budgets for coastal waters (Gordon *et al.* 1996) concentrate on the simplest case where an estuary or embayment is treated as a single box which is well-mixed both vertically and horizontally, and at steady-state. The Guidelines briefly describe approaches to treating systems with horizontal and/or vertical gradients in salinity, and encourage users to resolve temporal variation in loads and responses where data permit. However, it is not clear what errors might be incurred in failing to resolve spatial and temporal variation, or under what conditions these errors might be unacceptable.

Given that LOICZ wants to develop budgets for as many different coastal systems as possible, it is inevitable that budgets will be developed in systems with relatively sparse data, in which it is not possible in any case to resolve spatial and temporal variation. Even in systems where more data are available, it is not clear what level of spatial and temporal aggregation is desirable. While failure to resolve gradients may introduce systematic bias, attempts to over-resolve gradients may introduce high levels of noise. (In this recent Workshop, some participants aggregated data, and calculated budgets based on annual and system-wide averages, while others attempted budgets with fine spatial and temporal resolution.)

Choosing an "optimal" level of spatial and temporal resolution involves a complex mix of deterministic and statistical issues, and developing a generic, quantitative basis for making this choice will not be straightforward. Here, we make a start by examining the systematic errors involved in neglecting (i.e., averaging) spatial and temporal variation in some types of estuary. We consider in particular the effect of averaging temporal variation in simple well-mixed single-box systems. We consider also the effect of ignoring spatial variation in vertically well-mixed, horizontally varying (1-D, 1-layer) estuaries, and the effect of treating an estuary with a two-layer circulation as a single-box system.

1. Effect of Temporal Averaging Procedure

LOICZ budgets for estuaries are often calculated from measurements representing mean values of freshwater inflows, salinities, and concentrations (Gordon *et al.* 1996). However, in many of the systems the seasonal variations in these properties may be considerable. For example, in Lake Victoria (an estuary in south-eastern Australia; see Section 2.4) the freshwater inflows during the low flow season in summer/autumn 1989 were mostly less than 1000 ML d^{-1}, whereas winter flows and flows in the following spring were mostly greater than 6000 ML d^{-1}. Salinities responded accordingly. Low discharge salinities were about 25 ppt, but these reduced to less than 15 ppt during the high discharge periods. Here, we consider the effects of assuming that fluxes and exchange coefficients can be calculated from mean yearly values of flows, salinities and concentrations, rather than from an analysis that accounts for the variability of the system.

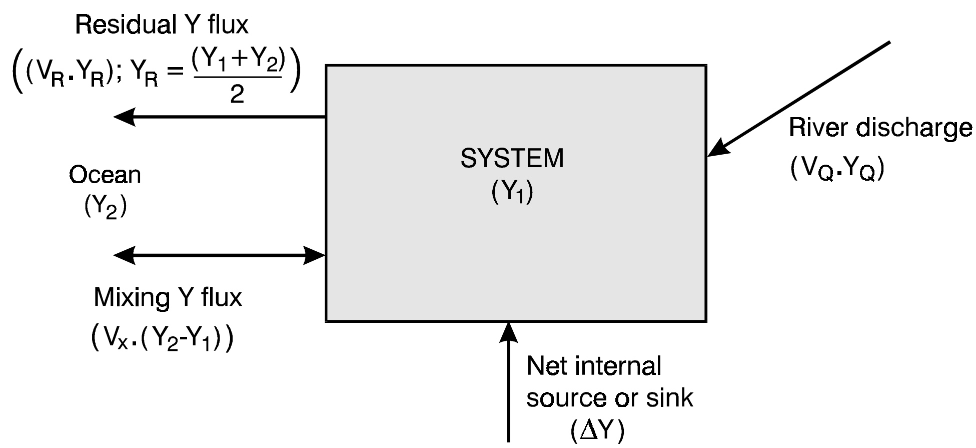

Figure III.1 Schematic of a single-box estuary.

For simplicity, we consider a single-box estuary that exchanges with the sea (Figure III.1). The nomenclature follows that in the LOICZ guidelines (Gordon *et al.* 1996). The estuary has a freshwater inflow, V_Q, which equals the net water exchange with the sea; that is, we assume zero gains or losses due to evaporation, precipitation, groundwater or to another water source within the estuary. Hence, the residual flow to the sea is $V_R = -V_Q$. A mixing exchange occurs with the sea due to winds, tides, or estuarine flow, which is characterised by the exchange velocity V_x. Note that in this analysis, we assume that flows and material transport into an estuary compartment are positive and that outflows are negative. Thus, V_R as depicted in Figure III.1 is negative.

Our hypothetical estuary is subject to two inflow regimes: one of low inflow for part of the year, and the second of elevated inflow for the remainder of the year. The system will be considered to be in quasi steady-state; that is, all flows and concentrations will be time invariant (and in balance) except at the instant that the flow is adjusted to a new level. With these assumptions, the equation for mass conservation in the estuary of a substance with concentration, Y, is:

$$Y_Q V_Q + (Y_2 - Y_1)V_x - \frac{Y_1 + Y_2}{2} V_Q + \Delta Y = 0 \qquad (1.1)$$

In this equation, Y_Q, Y_1, and Y_2 are the concentrations in the river inflow, in the estuary and in the sea respectively. The net internal source or sink of the substance within the estuary is ΔY. In Figure III.1, this source/sink term has been schematised as an internal gain into the water column through the bottom (positive).

This equation can be solved for Y_1 as:

$$Y_1 = \frac{2Y_Q V_Q + Y_2(2V_x - V_Q) + 2\Delta Y}{2V_x + V_Q} \tag{1.2}$$

If the substance is salt, then we can use Eq.1.1 to estimate V_x from measured salt concentrations in the estuary and in the sea. For salt, Y is S, $S_Q = 0$, and $\Delta S = 0$, so:

$$S_1 = \frac{2V_x - V_Q}{2V_x + V_Q} S_2 \tag{1.3}$$

Let the low-flow regime into the estuary have inflow, V_Q^0 and let it pertain for fraction, θ, of the year. The high flow regime has inflow, V_Q^+, and applies for the rest of the year, $(1-\theta)$. We shall specify that the high inflow is R times larger than the low inflow; that is:

$$V_Q^+ = RV_Q^0 \tag{1.4}$$

and that V_x remains constant through the year and is expressed in terms of V_Q^0 as:

$$V_x = rV_Q^0 \tag{1.5}$$

The assumption of constant V_x is not generally justifiable and is undertaken for simplicity and for illustrative purposes. One might expect that for very small inflows, estuarine circulation would be weak, and that mixing exchange would be dominated by winds and tides. For intermediate inflows, the estuarine circulation would become relatively stronger, but at very high inflows the estuarine circulation would again weaken as salt is flushed out of the estuary almost completely.

The salinity and other concentrations within the estuary will differ between the low- and high-inflow regimes. From Eqs. 1.3-1.5, the salinities in the low and high-flow regimes are calculated as:

$$S_1^0 = \frac{2V_x - V_Q^0}{2V_x + V_Q^0} S_2 = \frac{2r-1}{2r+1} S_2 \tag{1.6}$$

$$S_1^+ = \frac{2V_x - V_Q^+}{2V_x + V_Q^+} S_2 = \frac{2r-R}{2r+R} S_2 \tag{1.7}$$

The ocean salinity, S_2, has been assumed to be constant through the year. In effect, S_1^0 and S_1^+ are the estuarine salinities that would occur for our assumed flow regimes, ocean salinity, and exchange flow speeds.

The average salinity through the year is calculated from Eqs. 1.6 and 1.7 as:

$$\begin{aligned}\overline{S}_1 &= \theta S_1^0 + (1-\theta)S_1^+ \\ &= \alpha S_2\end{aligned} \qquad (1.8)$$

where:

$$\alpha = \theta\frac{2r-1}{2r+1} + (1-\theta)\frac{2r-R}{2r+R} \qquad (1.9)$$

Similarly, the yearly averaged inflow into the estuary is:

$$\begin{aligned}\overline{V}_Q &= V_Q^0 + (1-\theta)V_Q^+ \\ &= \beta V_Q^0\end{aligned} \qquad (1.10)$$

with:

$$\beta = \theta + (1-\theta)R \qquad (1.11)$$

As is most often done, we use the average values of the estuarine salinity and inflow to calculate the estuarine mixing flow. From Eq. 1.1, the mass balance equation for salt is written as:

$$(S_2 - \overline{S}_1)\overline{V}_x - \frac{\overline{S}_1 + S_2}{2}\overline{V}_Q = 0 \qquad (1.12)$$

from which:

$$\overline{V}_x = \frac{\overline{S}_1 + S_2}{2(S_2 - S_1)}\overline{V}_Q \qquad (1.13)$$

Substitution for $\overline{S}_1$ and $\overline{V}_Q$ from Eqs. 1.8 and 1.10 gives:

$$\overline{V}_x = \gamma V_Q^0 \qquad (1.14)$$

where:

$$\gamma = \frac{\alpha\beta + \beta}{2(1-\alpha)} \qquad (1.15)$$

However, the 'true' V_x is equal to rV_Q^0, but in general $\gamma \neq r$. Averaging salinities and inflows for calculating V_x results in an error in estimating this mixing flow.

The salinity budget allows calculation of V_x, but we really want to know the effects of averaging on the estimation of fluxes of substances other than salt. For simplicity, we shall assume that the concentration of the substance in the inflow is constant through the year. Thus, the input load to the estuary is proportional to the magnitude of the inflow. We shall also assume that the marine concentration of the substance being considered is low enough that we can set $Y_2 = 0$. With these assumptions, Eq. 1.1 becomes:

$$Y_1 = \frac{2Y_Q V_Q + 2\Delta Y}{2V_x + V_Q} \tag{1.16}$$

The internal gain term is assumed to be constant and is expressed as a fraction, ϕ, of the input load under low flow conditions; that is:

$$\Delta Y = \phi Y_Q V_Q^0 \tag{1.17}$$

Using Eqs. 1.4, 1.5, 1.16 and 1.17, we calculate Y_1 under low and high inflow conditions in analogous fashion to the calculation of S_1^0 and S_1^+ using Eqs.1.6 and 1.7:

$$Y_1^0 = 2Y_Q \frac{1+\phi}{2r+1} \tag{1.18}$$

$$Y_1^+ = 2Y_Q \frac{R+\phi}{2r+R} \tag{1.19}$$

Averaging the concentration over the year gives:

$$\begin{aligned} \overline{Y_1} &= \theta Y_1^0 + (1-\theta)Y_1^+ \\ &= \psi Y_Q \end{aligned} \tag{1.20}$$

where:

$$\psi = 2\theta \frac{1+\phi}{2r+1} + 2(1-\theta)\frac{R+\phi}{2r+R} \tag{1.21}$$

As we did using average salinity and inflow to calculate V_x, we can use the average inflow, $\overline{V}_x$, and estuarine concentration to calculate the average flux. Rearrangement of Eq. 1.1 gives:

$$\overline{\Delta Y} = -Y_Q \overline{V_Q} + \overline{Y_1}\overline{V_x} + \frac{\overline{Y_1}\overline{V_Q}}{2} \tag{1.22}$$

After substituting for $\overline{V}_Q$ (Eq. 1.10), $\overline{Y}_1$ (Eq. 1.20), and $\overline{V}_x$ (Eq. 1.13), we obtain:

$$\overline{\Delta Y} = (-\beta + \gamma\psi + \frac{\beta\psi}{2}) Y_Q V_Q^0 \tag{1.23}$$

The 'true' flux is $\overline{\Delta Y} = \phi Y_Q V_Q^0$ so that the relative size of the estimated flux to the true flux is:

$$\begin{aligned} \frac{\overline{\Delta Y}}{\Delta Y} &= (-\beta + \gamma\psi + \beta\psi / 2) / \phi \\ &= \frac{(\theta + R - \theta R)(\theta R + 2r + 1 - \theta)}{2rR + R + 2\theta R - 2\theta r R} \end{aligned} \tag{1.24}$$

after substitution for β, γ, and ψ. Note that $\overline{\Delta Y} / \Delta Y$ is not a function of the size of the internal gain term.

Evaluation of Eq. 1.24 shows that $\overline{\Delta Y} / \Delta Y$ has its largest values when $\theta \sim 0.5$; that is, when the low- and high-inflow regimes are of similar period. When the flow regime approaches the limits of being all low inflow ($\theta = 1$) or all high inflow ($\theta = 0$), the error in the estimation of ΔY approaches zero. Figure III.2 shows $\overline{\Delta Y} / \Delta Y$ as a function of r and R when $\theta = 0.5$. Thus, the Figure shows approximately the largest error that is obtained due to temporal averaging. The results have been plotted for $r > R/2$ since for

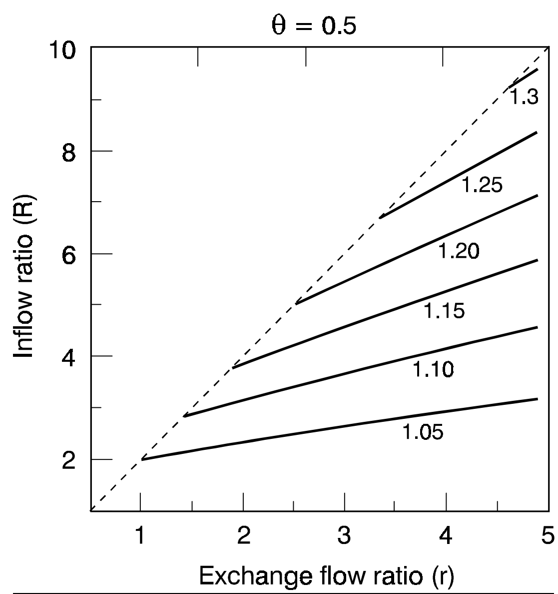

Figure III.2 Contours of $\overline{\Delta Y} / \Delta Y$ plotted versus the ratio of inflow magnitudes and the exchange flow ratio for $\theta = 0.5$. (Zone above dashed line is physically unreasonable.)

smaller values of R, S_1^+ would be negative (Eq. 1.7) which is physically impossible. The relative error increases as r decreases and as R increases; that is, the error is largest for estuaries that are still relatively fresh during the time of low inflow and for those subject to a large change in river inflow through the year. When r = 1.25, which is the value calculated when $S_1 = 0.5S_2$, the over-estimation of ΔY is 8% at R = 2.5, its maximum value. For $S_1 = 0.8S_2$, $r = 4.5$ and the over-estimation of ΔY reduces to 3% at R = 2.5. For this value of r, the maximum over-estimation of ΔY is 30% which occurs when R = 9.

The fundamental problem with estimating fluxes using Eq.1.22 lies in the invalidity of the averaging used to form this equation from Eq. 1.1. Rather than using Eq.1.22 to estimate the internal gain/loss, we should use:

$$\overline{\Delta Y} = -\overline{Y_Q V_Q} + \overline{Y_1 V_x} + \frac{\overline{Y_1 V_Q}}{2} \tag{1.25}$$

In this equation, the terms are the averages of the products of concentrations and flows rather than the products of the averages of these quantities.

2. Effects of Horizontal Averaging

We are interested in the effects on the calculated budgets (and the estimated internal sources and sinks in particular) of averaging along spatial gradients in estuaries which are vertically and transversely well-mixed, but have classical long-estuary mixing gradients of salinity and other tracers. We assume that the estuary is at steady-state, and that salinity (*S)* and another tracer (*Y)* are functions of the long-estuary coordinate, *x*; that is, $S = S(x)$ and $Y = Y(x)$. Also, we assume that enough samples of S and Y are taken to compute volume-weighted averages $S_1 = \langle S \rangle$ and $Y_1 = \langle Y \rangle$ for the estuary. Suppose the fresh-water runoff into the head of the estuary is V_Q, that the marine salinity is S_2 as in Figure III.1, and that the marine concentration of tracer Y is zero ($Y_2 = 0$).

Applying the LOICZ Guidelines, we would compute a mixing exchange V_x between the estuary and the ocean given by:

$$V_x = \frac{\langle S \rangle + S_2}{2(S_2 - \langle S \rangle)} V_Q \tag{2.1}$$

by analogy with Eq. 1.13.

We then calculate a net export of tracer Y from the estuary to be:

$$F_Y = \frac{\langle Y \rangle V_Q}{2} + \langle Y \rangle V_x \tag{2.2}$$

Given a known load of Y, L_Y, we would calculate an internal source ΔY given by:

$$\Delta Y = F_Y - L_Y \tag{2.3}$$

The question here is how the calculated export relates to the real export, and how this affects the estimated internal source. This obviously depends on the mixing and cycling of Y within the estuary. We consider here the special case where the tracer Y is conservative, and the load of Y is all due to a single-point source of Y at location x^*, and salinity S^*.

We can think of this special case in two ways. In many urban estuaries, there is a large point source of nitrogen and phosphorus located at some point along the estuary. This analysis then shows directly the bias that would be incurred by trying to estimate the fate of N and P from that source while ignoring the long-estuary gradient and source location. However, the analysis has a broader interpretation. In principle, we can describe the dynamics of any non-conservative tracer as corresponding to that of a conservative tracer with a source term $Q(x)$ distributed along the estuary. A LOICZ budget based on averaged concentrations will produce an estimate of the total source ΔY, which is a weighted average $\int wQ\,\mathrm{d}x$. The analysis here shows how the weight $w(x)$ varies along the estuary.

For a conservative tracer, Y, in a one-dimensional estuary, it can be shown (Officer 1979) that the downstream flux of Y at any point x and salinity S is given by:

$$F(x) = V_Q(Y - S\frac{\mathrm{d}Y}{\mathrm{d}S}) \tag{2.4}$$

Given the load L_Y at point x^*, salinity S^*, it follows from this equation that downstream of x^*, $F_Y = L_Y$ (constant), and:

$$Y = \frac{L_Y(1 - S/S_2)}{V_Q} \tag{2.5}$$

Upstream of x^*, $F_Y = 0$, and:

$$Y = \frac{L_Y S(1 - S^*/S_2)}{S^* V_Q} \tag{2.6}$$

Examples of these concentration distributions are shown in Figure III.3. Note that the tracer concentration downstream of the source is independent of the source location, but the peak tracer concentration, $(L_Y/V_Q)(1 - S^*/S_2)$, declines to zero as the point source is located further downstream and S^* approaches S_2. This reflects the increase in "effective" flushing rate as we approach the mouth of the estuary.

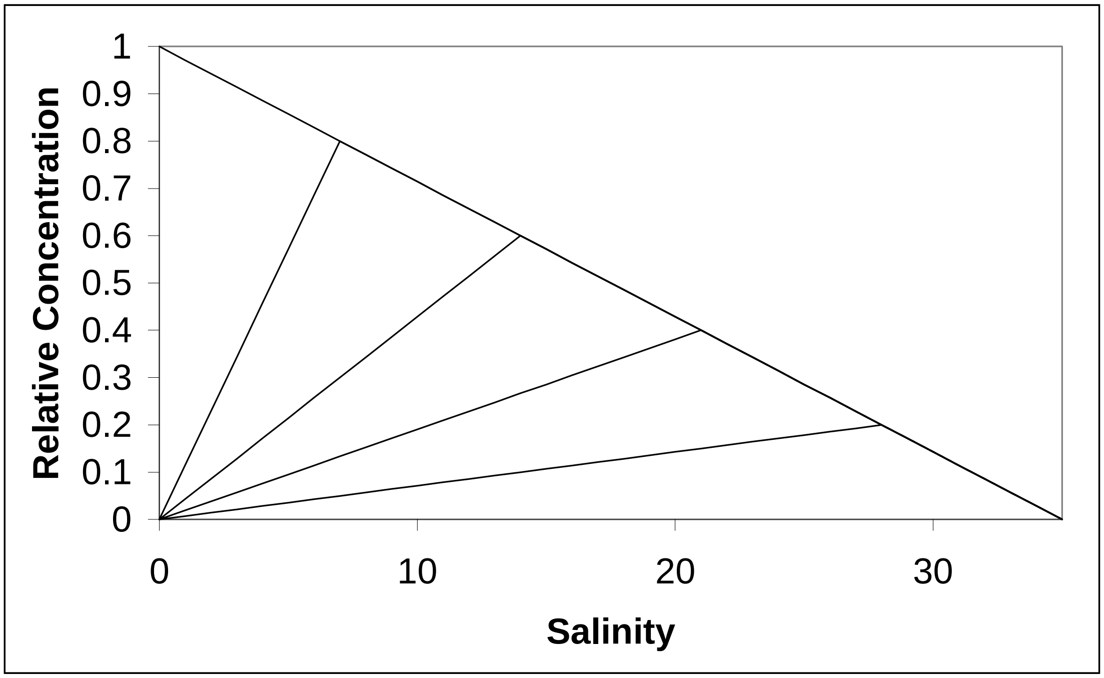

Figure III.3 Relative concentration vs salinity for a conservative tracer with fixed load located at salinity *S = 0, 7, 14, 21, 28.**

It is obvious from Figure III.3 that the average concentration, $<Y>$, diminishes as the source is located closer to the mouth. Because $<Y>$ is a volume-weighted average, we need to make further assumptions about the distribution of $S(x)$ and cross-section area $A(x)$ in order to compute $<Y>$ as a function of x^*. We consider two cases. The first assumes a channel with constant cross-section and a linear increase in S with x, so that a volume-weighted average is the same as a salinity-weighted average. Then, from integration of Eqs. 2.5 and 2.6:

$$<Y>= \frac{L_Y(1-S^*/S_2)}{2V_Q} \qquad (2.7)$$

As $<S>= S_2/2$, $V_x = 1.5V_Q$ (from Eq. 2.1), and the LOICZ (single-box) export is:

$$F_Y = L_Y(1-S^*/S_2) \qquad (2.8)$$

so:

$$\Delta Y = -\frac{L_Y S^*}{S_2} \qquad (2.9)$$

In other words, if a known external load of a conservative tracer occurs at the head of the estuary, the single-box (spatially-averaged) budget will correctly estimate the export, and correctly conclude that there is no internal sink. However, if this load is discharged at some point along the estuary with salinity S^*, the spatially-averaged budget will underestimate the export by a factor $(1-S^*/S_2)$, and incorrectly conclude that there is a large internal sink. As the source location approaches the mouth of the estuary, this false sink will approach 100% of the load. If there is a natural sink or source within the estuary, this sink or source will be under-estimated in the budget by the same factor $(1-S^*/S_2)$. A natural sink, which is uniformly distributed along the length of the estuary, will be under-estimated by 50%.

The assumptions of constant channel width and linear decline in S with x are not very realistic. For a more realistic configuration, consider an estuary of length x_L, where the cross-section area increases linearly with x as $A=\mu x$. Suppose $\lambda = V_Q/\mu D$ where the long-estuary diffusivity, D ($m^2\ s^{-1}$), is constant. Then, it is possible to show that $S = S_2(x/x_L)^\lambda$, and:

$$< S >= \frac{S_2}{2(\lambda+2)} \tag{2.10}$$

and:

$$< Y >= \frac{L_Y \lambda (1-S^{*2}/S_2{}^2)}{V_Q(\lambda+2)} \tag{2.11}$$

Applying a single-box LOICZ budget gives:

$$F_Y = L_Y(1-S^{*2}/S_2{}^2) \tag{2.12}$$

Once again, if the source is located downstream of the head of the estuary, the single-box budget underestimates the export, and will incorrectly conclude that there is an internal sink. In this case, the error is smaller for a given value of S^* but still approaches 100% as S^* approaches S_2. Alternatively, if there is a natural internal sink distributed uniformly throughout the estuary which takes up Y at a fixed rate per unit area of bottom, then the single-box budget will underestimate this sink by a factor $\lambda/(\lambda+1)$.

These errors are potentially large enough to badly distort budgets and, where there are large point sources located along the estuary, will render any estimates of internal sources or sinks meaningless. The examples are idealised, especially in the assumption that there are sufficient data to compute volume-weighted averages. In a more typical case where a single-box budget is calculated, there may be only a few measurements over the length of the estuary. Budgets are then subject to additional uncertainty depending on how these observations sample the long-estuary gradients.

If there are sufficient data to resolve long-estuary gradients, the bias involved in single-box budgets can in principle be avoided by computing spatially-resolved budgets. However, this may introduce other kinds of errors. According to Officer's result above, the net export from the mouth of the estuary is controlled by the gradient $\mathrm{d}Y/\mathrm{d}S$, at the mouth of the estuary. However, long-estuary gradients of Y and S may be weak near the mouth, and subject to considerable local spatial and temporal variability, making estimates of the gradient $\mathrm{d}Y/\mathrm{d}S$ highly uncertain. Caution must be exercised in choosing spatial compartments (i.e., salinity intervals) which are large enough to provide adequate signal-to-noise in flux estimates.

3. Effects of Vertical Averaging

Dyer (1973) has classified estuaries into three general types depending on salinity – highly stratified, vertically homogeneous, and partially mixed. Both the highly stratified and partially mixed estuaries are characterised by having a two-layer flow. An upper layer, which is relatively fresh, flows seaward and overlies a more saline lower layer. Mixing and entrainment across the boundary between the two layers causes the surface layer to become more saline as it flows towards the mouth. The salt transported towards the sea in the surface layer is replaced by a landward flow in the lower layer. Pritchard (1969) developed a model for transport in such a two-layer system that allowed for the estimation of entrainment and mixing between the layers from conservation of salt and water. Wulff and Stigebrandt (1989) used such an analysis to derive nutrient budgets for a series of three basins connected in series through the Baltic Sea starting with Bothnian Bay. An analysis extended by two additional basins to the Kattegat is described in Gordon *et al.* (1996).

<u>a) Two-layer case</u>

Here, we address the question of how treatment of an estuary as a single box differs from treatment of an estuary as having an upper and a lower layer, from the standpoint of budget calculations. The two-layer estuary we consider is schematised in Figure III.4. Exchange between the upper and lower layers occurs through the entrainment term $V_e.(Y_{1d} - Y_{1u})$ and through the mixing term $V_m.(Y_{1u} + Y_{1d})/2$, Y_{1u} and Y_{1d} being the concentrations in the upper and lower layers, respectively. Freshwater discharges into the upper layer at rate V_Q and there is assumed to be no internal loss or addition to the inflow in either layer. Thus the water budget is expressed as:

$$V_Q + V_u + V_d = 0 \qquad (3.1)$$

where V_u and V_d are the inflows/outflows to the upper and lower layers. The terms ΔY_u and ΔY_d are net internal sources or sinks of the substance within the upper and lower layers. As before, we define all material flows as being positive for flows into an estuary compartment except for the entrainment flow, V_e, which is positive for a flow from the lower to the upper layer.

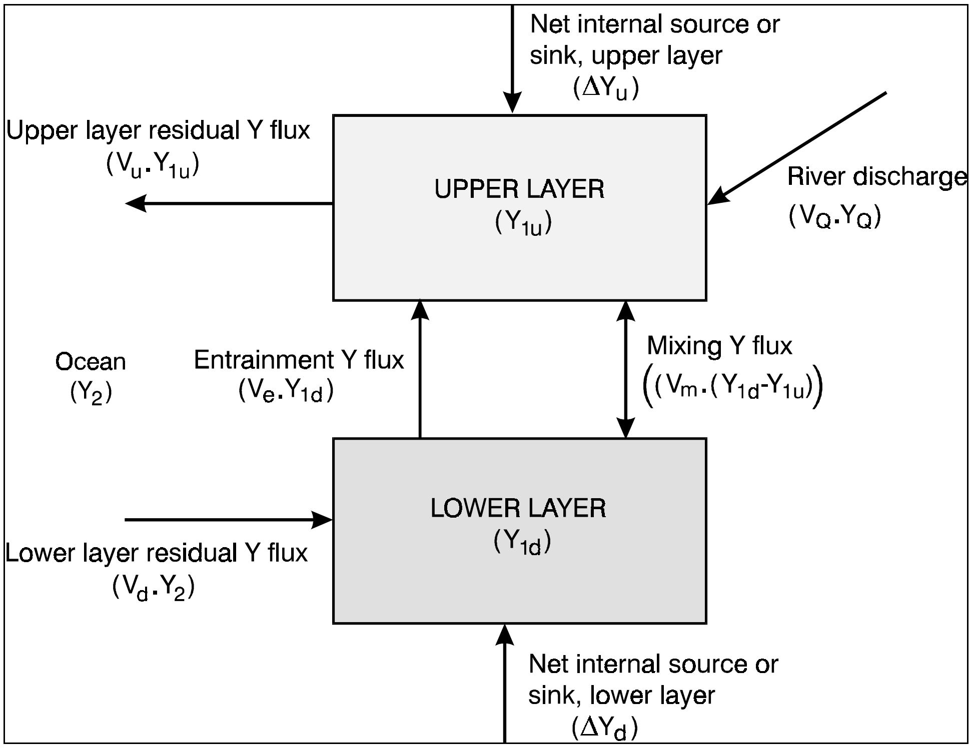

Figure III.4 Schematic of a two-layer estuary.

The material budgets for the upper and lower layers are:

$$Y_Q V_Q + Y_{1u} V_u + Y_{1d} V_e + (Y_{1d} - Y_{1u}) V_m + \Delta Y_u = 0 \tag{3.2}$$

$$Y_2 V_d - Y_{1d} V_e - (Y_{1d} - Y_{1u}) V_m + \Delta Y_d = 0 \tag{3.3}$$

If we add Eqs. 3.2 and 3.3 and use Eq. 3.1 to eliminate V_d , we obtain:

$$Y_Q V_Q + Y_{1u} V_u + Y_2 (-V_Q - V_u) + \Delta Y_u + \Delta Y_d = 0 \tag{3.4}$$

As in section 1, we obtain an equation for the salt balance by setting Y to be S, $S_Q = 0$, and $\Delta S = 0$. Solving Eq. 3.4 for the advective outflow in the upper layer gives:

$$V_u = \frac{S_2}{S_{1u} - S_2} V_Q \tag{3.5}$$

It is convenient to define non-dimensional variables denoted with a prime using:

$$S' = \frac{S}{S_2};\ V' = \frac{V}{V_Q};\ Y' = \frac{Y}{Y_Q} \tag{3.6}$$

Thus:

$$V_u' = \frac{1}{S_{1u}' - 1} \tag{3.7}$$

For the calculation of internal gain/loss from the estuary, we will assume that $Y_2 = 0$ as in section 1. Upon rearrangement of Eq. 3.4 and with the use of the non-dimensionalisation expressions, Eq. 3.6, we obtain:

$$\Delta Y_u + \Delta Y_d = -(1 + Y_{1u}' V_u') Y_Q V_Q \tag{3.8}$$

<u>b) Single-box case</u>

The case of a single box has already been considered in section 1, and we follow the analysis developed there. For the two-layer estuary, we have assumed that the salinities in the two layers are S_{1u} and S_{1d} (Figure 1). The equivalent salinity for the single-box estuary (assuming that the two layers have equal volume) would be the average of the salinities in the two layers; that is:

$$S_1 = \frac{S_{1u} + S_{1d}}{2} \tag{3.9}$$

and similarly the equivalent single-box concentration would be:

$$Y_1 = \frac{Y_{1u} + Y_{1d}}{2} \tag{3.10}$$

For the single-box analysis presented here, the equations for the exchange flow and for the internal gain/loss follow are the same as Eqs. 1.13 and 1.22 without the temporal averaging operators. The present case is time invariant. Thus:

$$V_x = \frac{S_1 + S_2}{2(S_2 - S_1)} V_Q \tag{3.11}$$

or, with Eq. 3.9, we obtain:

$$V_x = \frac{(S_{1u} + S_{1d})/2 + S_2}{2(S_2 - (S_{1u} + S_{1d})/2)} V_Q \tag{3.12}$$

Substitution of the non-dimensionalisation expressions, Eq. 3.6, yields:

$$V_x' = \frac{S_{1u}' + S_{1d}' + 2}{2(2 - S_{1u}' - S_{1d}')} \tag{3.13}$$

For the steady-state system we consider here, the internal gain/loss for a single-box estuary is obtained by rearranging Eq. 1.1 as:

$$\Delta Y = -Y_Q V_Q + Y_1 V_x + \frac{Y_1 V_Q}{2} \tag{3.14}$$

This equation is the same as Eqs. 1.22 and 1.25 without the temporal averaging overbar. Substitution of Eq. 3.10 for Y_1 and use of Eq. 3.6 to non-dimensionalize the variables allows calculation of the internal gain/loss for the single-box equivalent of a two-layer estuary to be:

$$\Delta Y = (-1 + (Y'_{1u} + Y'_{1d})(1/4 + V'_x/2))Y_Q V_Q \tag{3.15}$$

c) Comparison of cases

Figure III.5 shows the total internal gain/loss as functions of the upper and lower layer concentrations of substance Y for the single-box case (Eq. 3.15) non-dimensionalised by the input load, $Y_Q V_Q$ as:

$$\Delta Y'_1 = \frac{\Delta Y}{Y_Q V_Q} \tag{3.16}$$

The results have been shown for an upper layer salinity of $S'_{1u} = 0.25$ and a lower layer salinity of $S'_{1d} = 0.75$. These would correspond to the case of a partially mixed estuary.

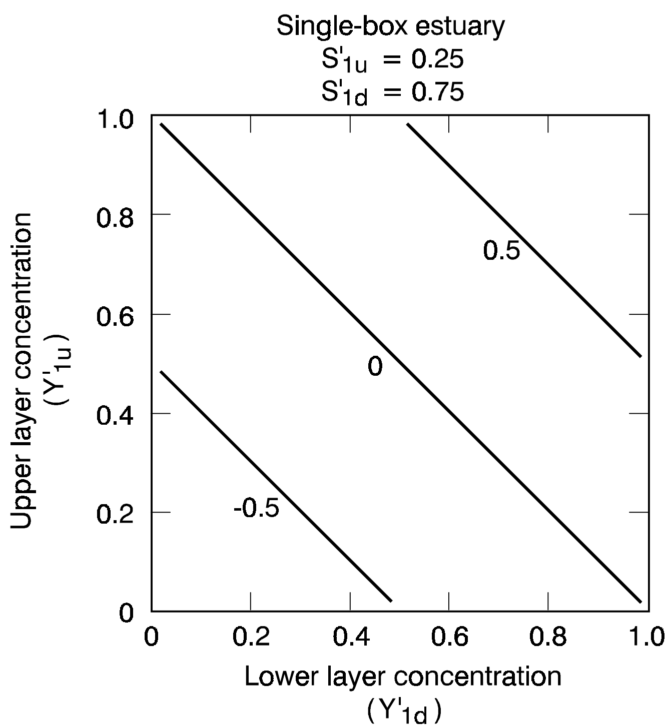

Figure III.5 Contours of ΔY_1 plotted against non-dimensional concentrations in upper and lower layer for the single-box estuary.

Of course, for the single-box estuary, the exchange to the sea and the calculation of internal gain/loss depend only on the average of the upper and lower layer salinities and concentrations. Accordingly, the isolines of the internal gain/loss function are the lines of constant $Y_{1u} + Y_{1d}$. For the chosen salinities, $\Delta Y_1'$ is negative when the estuarine concentration is less than the concentration in the inflow, but it is positive if the estuarine concentration is greater. The position of the isoline of zero internal gain depends on the average salinity. As the average salinity increases, $\Delta Y_1'$ also increases for fixed estuarine concentration.

Similarly, Figure III.6 shows the internal gain/loss for the two-layer case (Eq. 3.8) non-dimensionalised in the same way:

$$\Delta Y_2' = \frac{\Delta Y_u + \Delta Y_d}{Y_Q V_Q} \tag{3.17}$$

Clearly, the isolines of internal gain/loss for the two-layer case are very different from those determined for the single-box estuary. For the two-layer case, the isolines are horizontal, whereas for the single-box case they are diagonal. This result demonstrates that, even for the same salinity and concentration assumptions being made, the calculated internal gain/loss is critically dependent on the configuration chosen to represent the estuary.

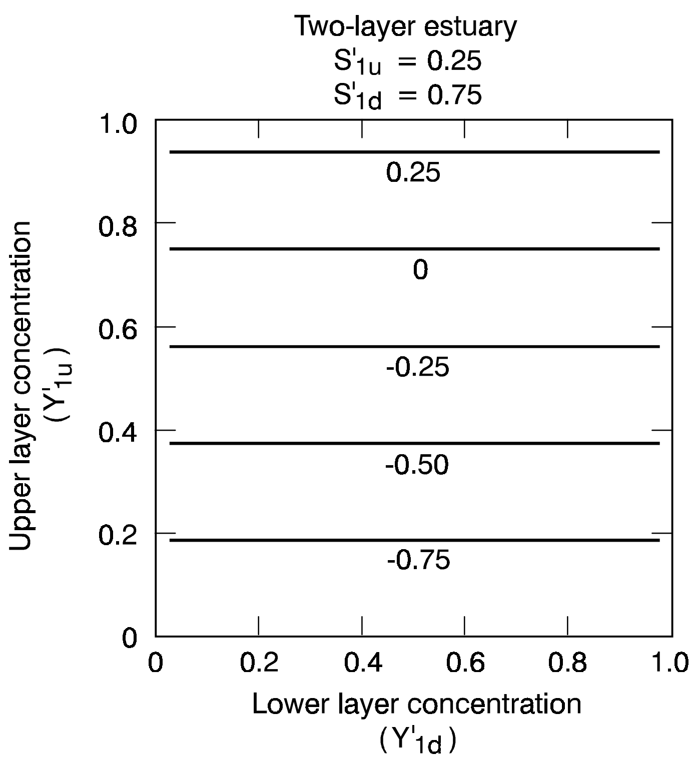

Figure III.6 Contours of ΔY_1 plotted against non-dimensional concentrations in upper and lower layer for the two-layer estuary.

As an example, suppose the upper layer salinity (non-dimensional) is 0.25 and the lower layer salinity is 0.75. Further suppose that the measured upper and lower layer concentrations are both 0.5. Then, Figure III.5 tells us that the inferred internal loss would be 0 using the single-box analysis. Howver, Figure III.6 tells us that internal loss would be about 0.3 ($\Delta Y_2'$ ~ -0.3) if the calculation were obtained using the two-layer representation.

We have already pointed out that for the single-box estuary the internal gain/loss is a function of the sum of the upper and lower layer concentrations. However, the two-layer case has the internal gain/loss to be a function of the salinity and concentration in the upper layer only (Eqs. 3.7 and 3.8). Of course, if the concentration of substance (not salt) is non-zero in the sea, then that concentration will enter the calculation of internal gain/loss as well.

The difference in calculated internal gain/loss between the two representations of estuaries arises from different fundamental assumptions about how the along-estuary solute transport occurs. In the single-box estuary, the advective flow out of the estuary carries with it a concentration halfway between oceanic and estuarine concentration. In a real estuary, the concentration at the seaward boundary of the box would be such if horizontal mixing within the estuary were vigorous. In this situation, we would expect that vertical gradients of salt and other solutes would be small. Consequently, the single-box representation is likely to be most appropriate to vertically homogeneous estuaries. The fundamental mixing dynamic represented in the two-layer representation is quite different. The surface-layer flow out of the estuary has a concentration equal to that near the center of the estuary. In other words, there is an implicit assumption that the amount of dilution of the seaward-flowing surface layer by horizontal mixing with ocean water is small. The two-layer representation is likely to be best for highly stratified estuaries.

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APPENDIX IV BUDGETS ESTIMATES BY NON-LOICZ APPROACHES

APPENDIX IVA RICHMOND RIVER ESTUARY (NEW SOUTH WALES): INTER-ANNUAL BUDGETS FROM SEASONAL AND EVENT MONITORING; A MODIFIED LOICZ APPROACH.

L. McKee and B.D. Eyre

Study Area Description

The Richmond River estuary is a sub-tropical shallow bar built system (Figure IV.1; also Figure 1.1)

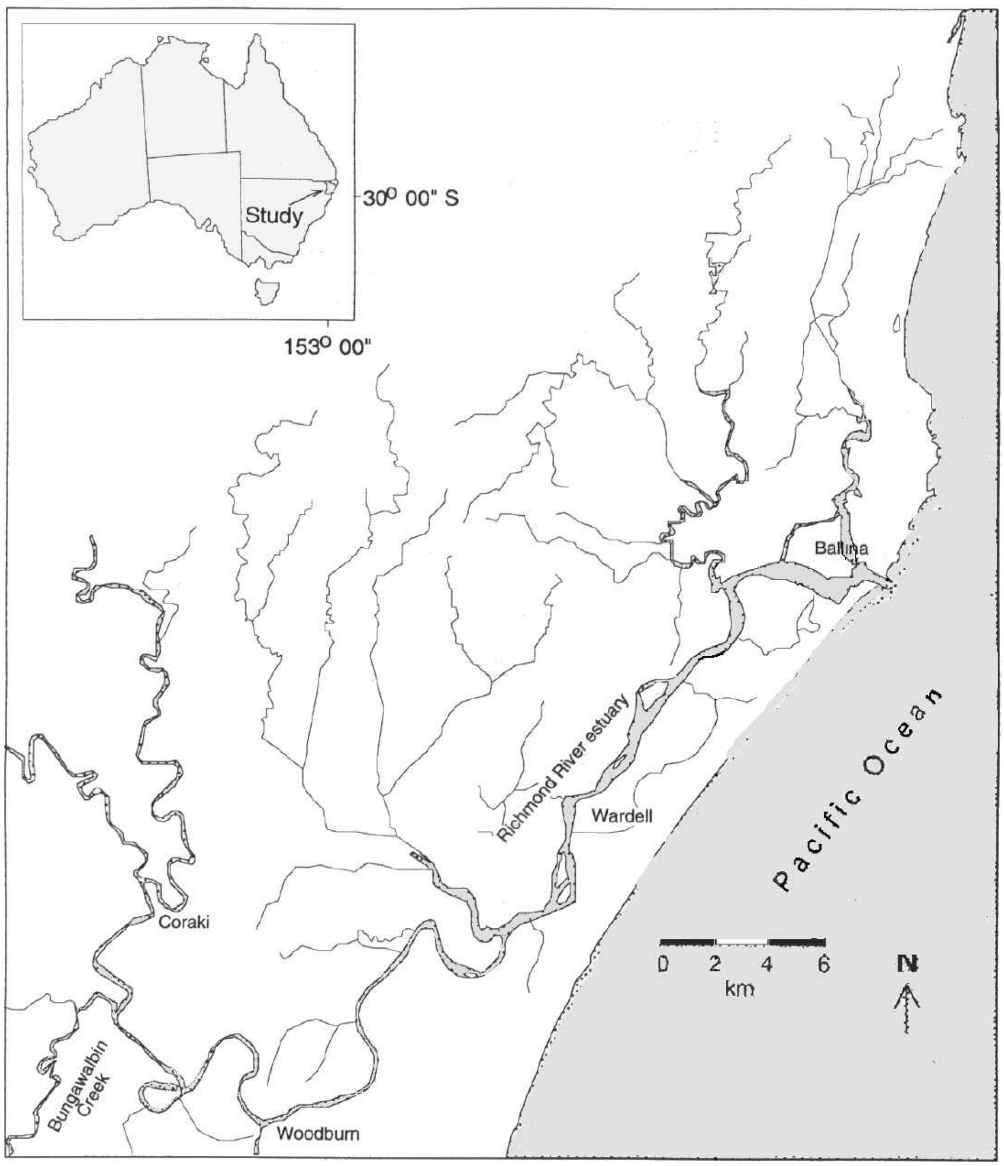


Figure IV.1 Richmond River estuary, New South Wales.

During the driest part of the year (September) the estuary is well mixed and ocean salt penetrates 40-50 km upstream. In the wet season the Richmond estuary can be flushed fresh to the mouth for up to several weeks each time heavy rain occurs in the catchment. The estuary has a surface area of 15 km^2, a surface catchment:estuary area ratio of 449, an average volume of about 54 million m^3, and a semi-diurnal tidal cycle. The tidal range varies from a minimum of 0.65 m on neap tides to a maximum of 1.9 m on spring tides. Average coastal rainfall varies from 1300 mm at Woodburn to 1800 mm at Ballina and rainfall has a seasonal late-summer dominated pattern. The estuary receives the majority of its freshwater discharge (annual average = 11.7 L s^{-1} km^{-2}) from the Richmond catchment above Woodburn which comprises 87% of the total catchment area (6861 km^2). Discharge during the study varied between 0.29 - 376 L s^{-1} km^{-2} resulting in estuarine flushing times from <1d to 176d. For the purpose of this study, the estuary is defined as the area downstream from Coraki (the maximum limit of salt intrusion during the dry season).

Sediment organic carbon ranges from 1% near Ballina to 14% in the upper reaches (Woodburn). The dominant producer in the system is unknown but is assumed to be phytoplankton. However, the lower reaches are lined by riparian mangroves and there are small areas of seagrasses in shallow protected areas of the lower estuary. The main land uses in the coastal plain include sugar cane, beef farming, urban, and scattered rural residential areas. The upper catchment, which supplies the majority of the freshwater to the system, has 41% forest cover, 53% dairy and beef grazing, and 6% cropping and horticulture. The average population density for the catchment is 14 persons km^{-2} of which 61% live in urban areas. There is one sewage treatment plant (STP) discharging into the lower Richmond River estuary at Ballina.

Methods

Diffuse loads

N and P concentrations derived from water samples collected on a flow-weighted basis from all terrestrial sources were combined with discharge on a monthly basis during low flow periods, and on an hourly basis during floods, using linear interpolation between samples. Loads from the upper catchment, coastal sub-catchment and cane land were added to give a total diffuse load to the estuary.

Atmospheric loads

N and P concentrations derived from precipitation samples gathered during rain events were combined with rainfall volume entering the estuary surface.

Sewage load

Nutrient loads via leaching from septic systems adjacent to the estuary were estimated using loads of TN (4 kg $person^{-1}$ yr^{-1}) and TP (1 kg $person^{-1}$ yr^{-1}) and applying these to the unsewered townships of Wardell and Broadwater adjacent to the estuary. The loads calculated are probably an over-estimate because it was assumed that all TN and TP from septic systems reached the estuary. The load of urban sewage from Ballina was estimated by integrating metered discharges and monthly average concentrations of nitrate, ammonia, and total phosphorus obtained from local authorities. Loads from leaching and treated sewage were combined to give a total sewage input.

Loads from urban runoff

Nitrogen loads associated with urban runoff were calculated by multiplying average TN concentration (1.4 mg L^{-1}) found in Lismore urban drains (the largest town in the Richmond River catchment) (C_{Urb}) by monthly rainfall at Lismore (R) and impervious area (A_{Imperv}), using a runoff coefficient of 50%, and summing from July to June for each year of the study:

$$Load(kgyr^{-1}) = \sum_{Jun}^{Jul} C_{Urb}\, 0.5 R A_{\mathrm{Imperv}}$$

Phosphorus loads were calculated similarly, using a TP concentration of 0.7 mgL^{-1}. Loads from the coastal towns of Ballina, Wardell, Broadwater, Woodburn, and Coraki were estimated by the ratio of each town's population to the population of Lismore, in the absence of nutrient concentration data for small towns adjacent to the estuary.

Estuarine 24 hour sampling

Water samples were collected at approximately 1.5 hour intervals over 23-26 hours during consecutive spring and neap tides at Ballina (Figure IV.1). Sampling was carried out over a range of catchment discharges on five occasions between July 1994 and June 1996. Water samples and velocity measurements were taken at three depths at three points across the estuary mouth (which has been modified to a rectangular cross-section for shipping purposes).

Each transect was echo-sounded to obtain cross-sectional area profiles. This area was adjusted for tidal fluctuations over each sampling period to give a cross-sectional area at a given time during each survey. Discharge for each transect was calculated by multiplying the average velocity in a vertical section (three sub-sections) by the sectional area (adjusted for tide height). Total discharge (m^3) was calculated by summing the sub-sections and multiplying by the time between samples (usually 1.5 hours). Nutrient loads for each tide at Ballina were calculated by multiplying the sample concentration (mg L^{-1}) by discharge (m^3) for each 1.5 hour period and summing over the full tidal cycle (25 hours).

Flood event sampling at the mouth

Sampling was undertaken on three occasions during flood events when the estuary was flushed fresh to the mouth at Ballina. Water samples were taken up to 4 times a day. Logistical difficulties did not allow the direct measurement of velocity. Instead, velocity and discharge were modelled (Hossain 1998), using the one-dimensional unsteady flow model (DUFLOW). Nutrient loads during the flood events were calculated, using the same methods described for tidal sampling. Mass loads for the rest of each month when a flood event occurred, were calculated by multiplying modelled discharge by the nutrient concentration of the last sample taken during flood event sampling.

Nutrient load verses discharge relationships for the estuary at Ballina

Loads calculated for months with storm discharge and loads derived from 24-hour sampling were used to develop rating relationships between catchment discharge and load. Variation in catchment discharge accounted for greater than 98% of the variation in nutrient loads. The rating relationships were used to predict the loads during periods when sampling was not undertaken.

Sediment nutrients

Samples were collected on three occasions for analysis of sediment nitrogen and phosphorus. The top 2-4 cm of sediment were grab-sampled from depositional areas below low tide level, near the edge of the estuary, at 13 locations. Loads of nutrients stored in the estuarine sediments were estimated by multiplying net sedimentation rates (Hossain 1998) for the 1994/95 year (5423 tonnes upper estuary, 12 497 tonnes lower estuary) and for the 1995/96 year (4664 tonnes upper estuary, 9256 tonnes lower estuary), by the average sediment nutrient composition (upper: 2.0 kg N t^{-1}, 0.9 kg P t^{-1}, lower: 0.9 kg N $tonne^{-1}$, 0.5 kg P $tonne^{-1}$).

Errors

Errors for each term in the nutrient budgets were relatively small when considered separately. However, the errors associated with the residual in the N budget incorporated the additive errors of all the other terms. Included were the errors associated with sediment and water column nutrient analysis, catchment discharge, calibration of the hydrodynamic model, and sedimentation rates. Unknown errors associated with the measurement of sewage loads, urban runoff, and precipitation were assigned the same error as that of the diffuse loads. Since these loading terms were small relative to ocean exchange and diffuse runoff, this approximation is of little consequence to the overall interpretation. The errors were then summed to give a total error for the residual of the TN budget. In the case of the stoichiometrically linked TDN and TDP budgets, the error assigned to the unknown term (*nfix-denit*) was calculated as the sum of the TDN errors and 16 times the sum of the TDP errors.

Results

Water Budget

The Richmond River estuarine water budget is driven mainly by catchment discharge (Figure IV.2). The majority (70-90%) of the annual catchment discharge can occur during short-lived events associated with tropical rain depressions. If rainfall minus evaporation over the estuary is assumed to be small (reasonable for an average year), the discharge through the mouth is assumed to be equivalent to the catchment discharge. The water budgets are presented simplistically (Figure IV.2); however, the catchment discharge and the estuarine water exchange through the mouth were quantified for the system using rainfall runoff relationships, a one-dimensional computer model and an hourly time step (Hossain 1998).

Total nitrogen budget

Annual total nitrogen budgets are presented where the single unknown term can be considered the net N_2 exchange with the atmosphere (Figure IV.3). Using a total

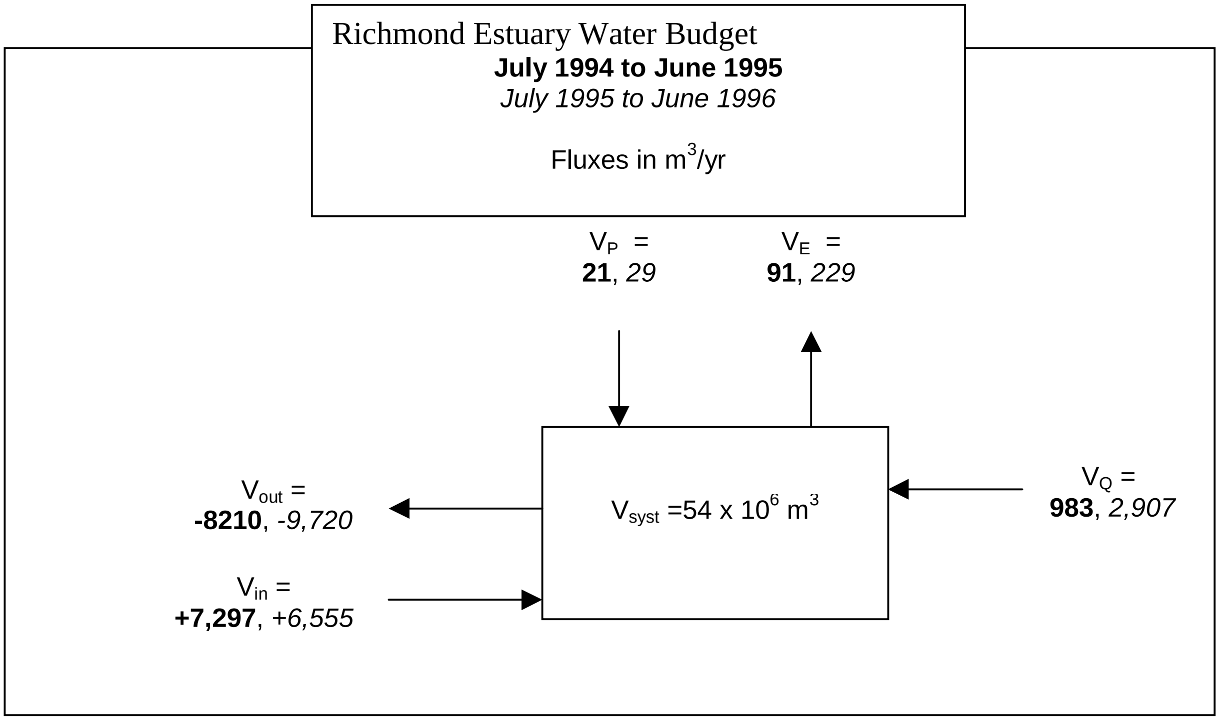


Figure IV.2 Water budget, Richmond River estuary.

nitrogen budget, (*nfix-denit*) rates computed for each year are not distinguishable from zero (Table IV.1). This occurs because the residuals for the annual budgets are small compared with the sum of errors associated with the quantification of the other terms. Therefore, in sub-tropical systems such as the Richmond River estuary, where much of the annual loads are transmitted conservatively through the estuary during wet season floods, annual N budgets are unlikely to allow the calculation of atmospheric N_2 exchange.

Table IV.1 Budget Comparisons, Richmond River estuary.

Budget	(*nfix-denit*) [Redfield Ratio] mmol m^{-2} d^{-1}	(*nfix-denit*) [sediments] mmol m^{-2} d^{-1}	(*p-r*) [Redfield Ratio] mmol m^{-2} d^{-1}	(*p-r*) [sediments] mmol m^{-2} d^{-1}
TN budget 94/95	-1.4 ± 3.5		---	---
TN budget 95/96	+0.2 ± 11.9			
TDN:TDP 94/95 dry	-5.0±1.0	-9.2 ± 1.6		
TDN:TDP 94/95 annual	-4.8 ± 2.5	-8.4 ± 4.8	-101	-33
TDN:TDP 95/96 dry	-3.1 ± 0.5	-6.7 ± 1.1	-94	-31
TDN:TDP 95/96 annual	+1.2 ± 9.9	-0.6 ± 15.9	-47	-15
Eyre 1996 (this issue)	---	+3.9	+8	---

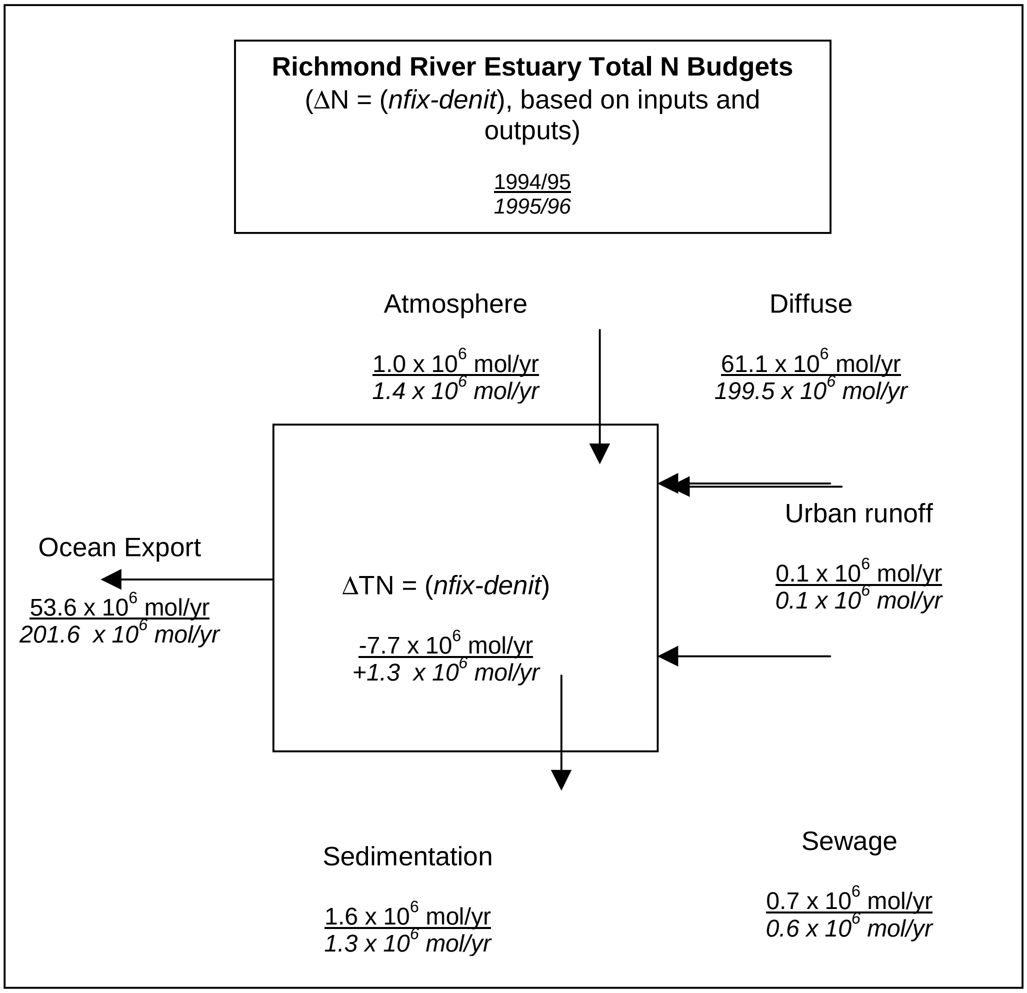
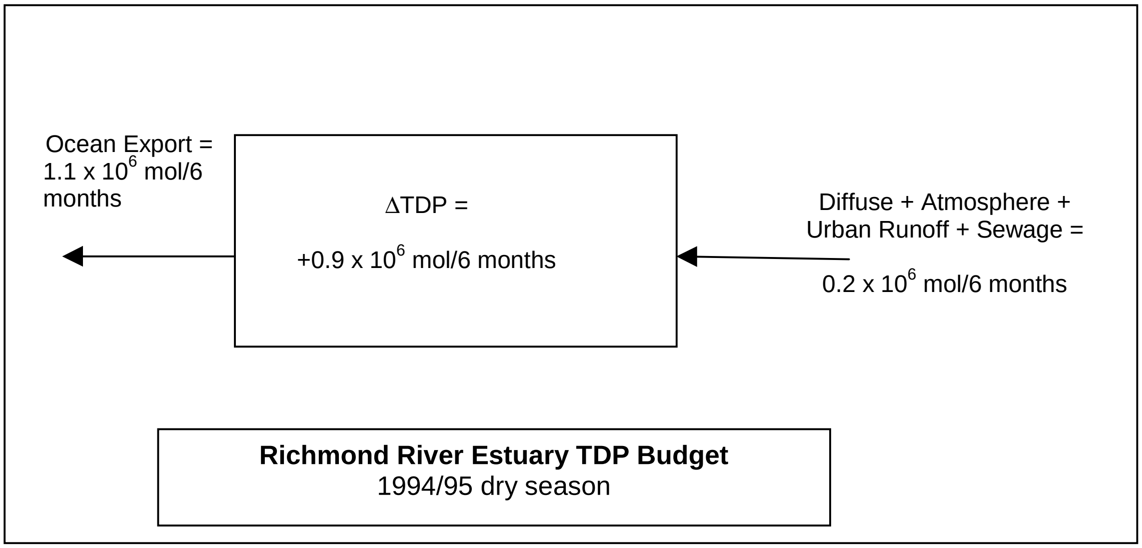


Figure IV.3 Total N budgets for two years, Richmond River estuary. [ΔTN is calculated as the difference between the summed inputs and summed outputs.]

TDP and TDN stoichiometry

Budgets for TDP and TDN are presented using both the sediment composition of Richmond sediments (323:3.2:1) and the Redfield ratio (106:16:1) for comparison (Figure IV.4 to IV.7). When (*nfix-denit*) is computed using stoichiometric relationships (approximately the LOICZ method) and annual TDP and TDN budgets, the errors render the results indistinguishable from zero for the same reasons as suggested for the TN budgets. If the dry seasons are considered separately, net denitrification computed for both years was significantly different from zero. During the dry seasons in the Richmond River estuary, flushing intervals are long relative to the biogeochemical processes occurring within the system. As such, the residuals of the TDP and TDN budgets are large relative to the errors associated with the quantification of the input and output terms. Another estimate of (*nfix-denit*) was made for the Richmond River estuary for 1996 (Eyre, Section 2.1.2), using the LOICZ methodology (Gordon *et al.* 1996), estimated net nitrogen fixation.



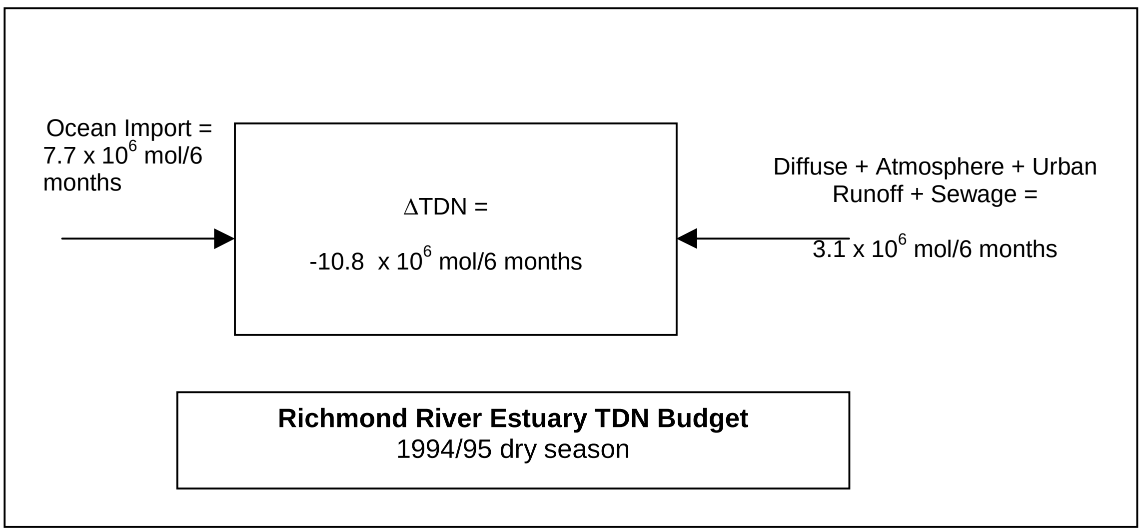
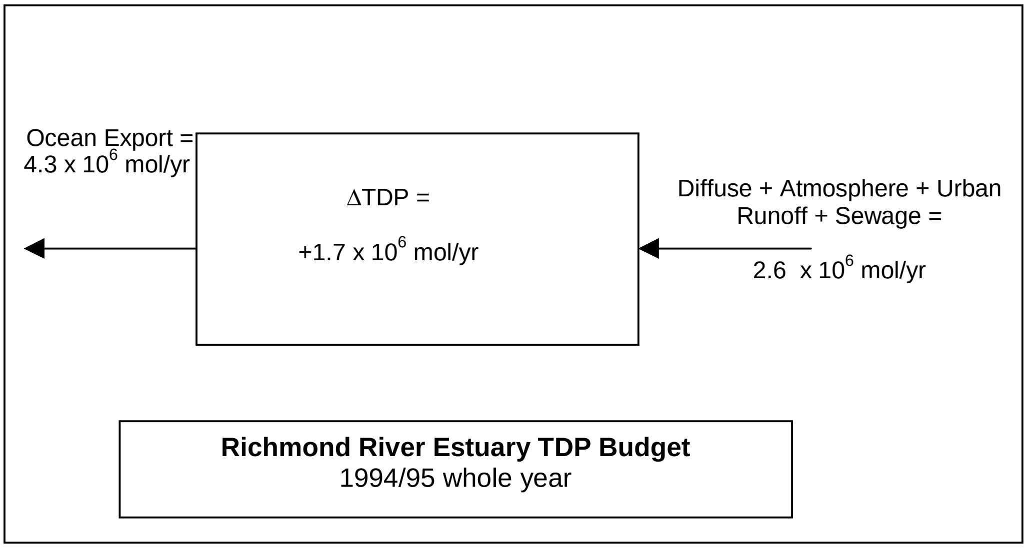


Figure IV.4 TDP and TDN budgets for the 1994/1995 dry season Richmond River estuary. [ΔTDP and ΔTDN are estimated by the difference between the summed inputs and oceanic output. The estimate of (*nfix-denit*) given in Table IV.1 is determined according to the LOICZ method ([*nfix-denit*] = ΔDIN_{obs} -ΔDIN_{exp}, where ΔDIN_{exp} is estimated either from the sediment particulate N:P ratio or from the Redfield N:P ratio)].



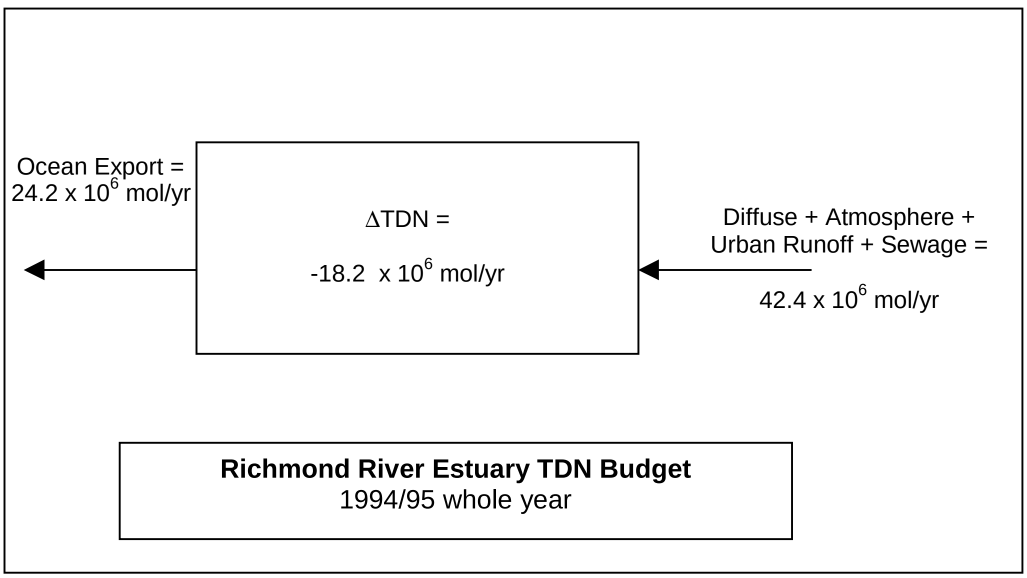
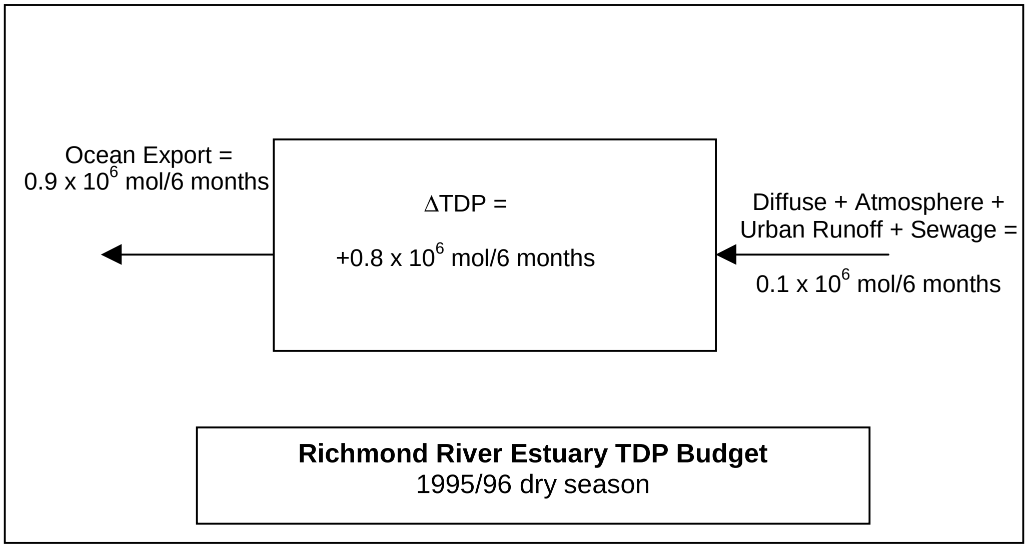


Figure IV.5 TDP and TDN budgets for the entire 1994/95 water year, Richmond River estuary. [Details as given in Figure IV.4.]



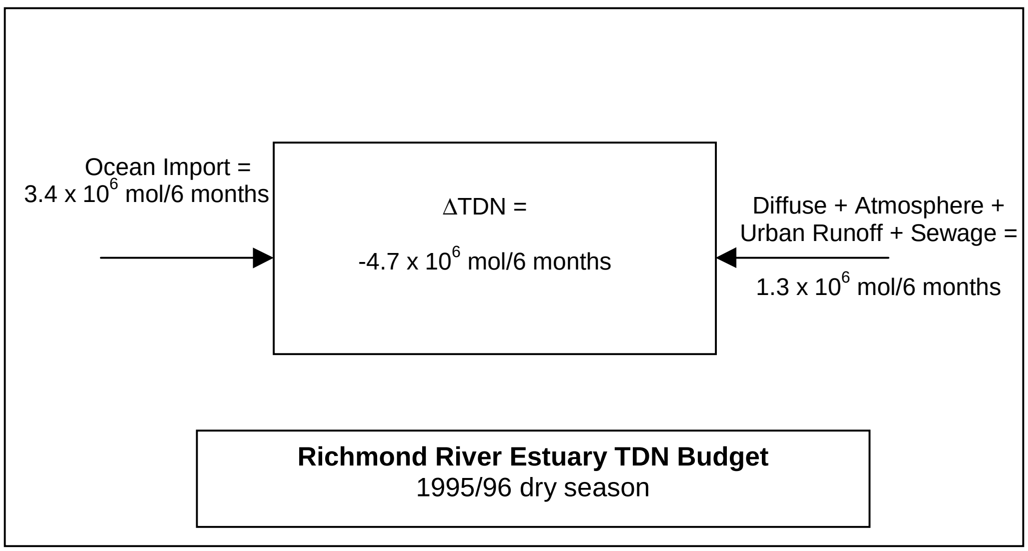
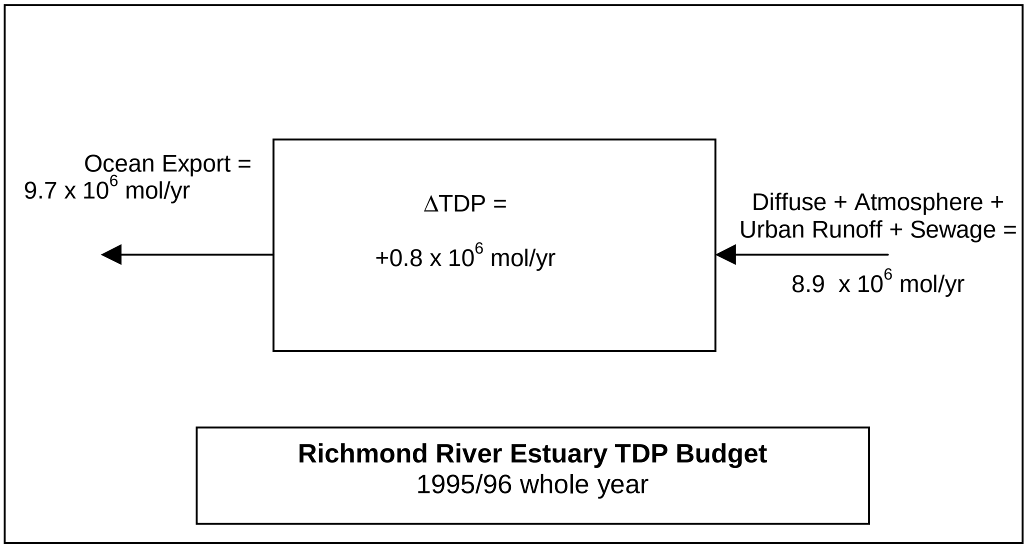


Figure IV.6 TDP and TDN budgets for the 1995/96 dry season, Richmond River estuary. [Details as given in Figure IV.4.]



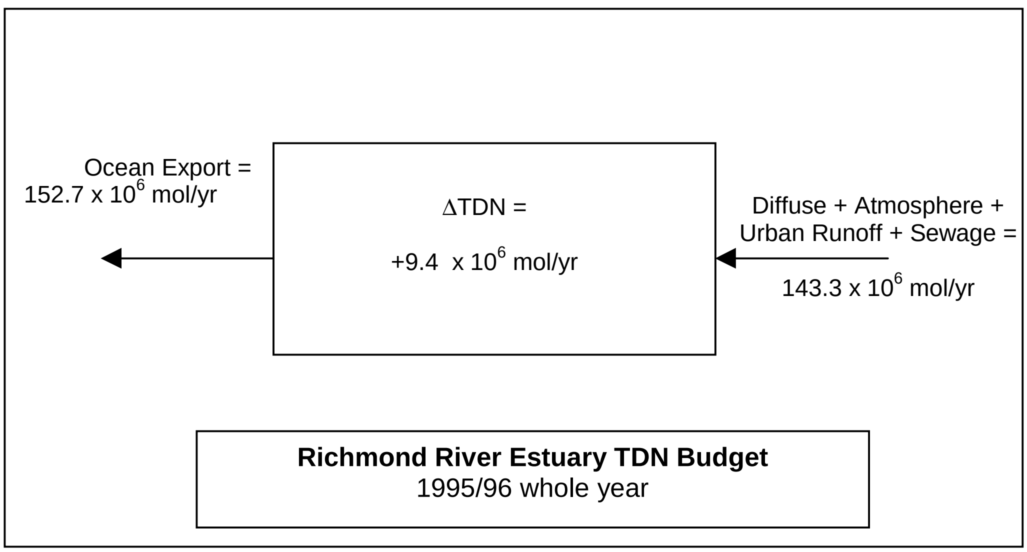


Figure IV.7 TDP and TDN budgets for the entire 1995/96 water year Richmond River estuary. [Details as given in Figure IV.4.]

Phosphorus-carbon stoichiometry

Scaling ΔTDP to the sediment C:P ratio (LOICZ methodology) gives an estimate of the net ecosystem metabolism (*p-r*) (Table IV-1). Here, the chosen sediment nutrient ratios make a large difference in the magnitude of (*p-r*). The Richmond River estuary appears to be respiring about 30 to 100 mmol m^{-2} d^{-1} organic carbon, depending on the chosen sediment nutrient ratios. These rates are larger than predicted by Eyre (Section 2.1.2), using the LOICZ budgeting approach, and opposite in sign.

APPENDIX IVB SWAN CANNING ESTUARY: SEASONAL NUTRIENT FLUXES USING A STEADY-STATE MASS BALANCE MODEL

D. Fredericks, D. Heggie and A. Longmore

Nutrient Mass Balance

Steady-state mass balances for estuaries have been considered in some detail by a number of authors (Liss and Spencer 1970, Boyle et al. 1974, Officer 1979, Smith and Atkinson 1983, Kaul and Froelich 1984, Smith and Veeh 1989) and despite a number of limitations have been applied in a wide variety of settings (Fisher *et al*. 1988, Zhang *et al.* 1997, Eyre and Twigg 1997, Smith and Atkinson 1983).

In these models the catchment flux of any species to the estuary is given by

$$F_c = QC_0$$

where F_C is the catchment flux, Q is the discharge and C_0 is the freshwater concentration of the species. The flux of this species past an isohaline surface isohaline surface (x) within the estuary is estimated from

$$F_x = Q(Ck\,*)$$

where *Ck** is the apparent freshwater concentration at the isohaline - the y intercept of the tangent to concentration salinity curve as illustrated in Figure IV.8.

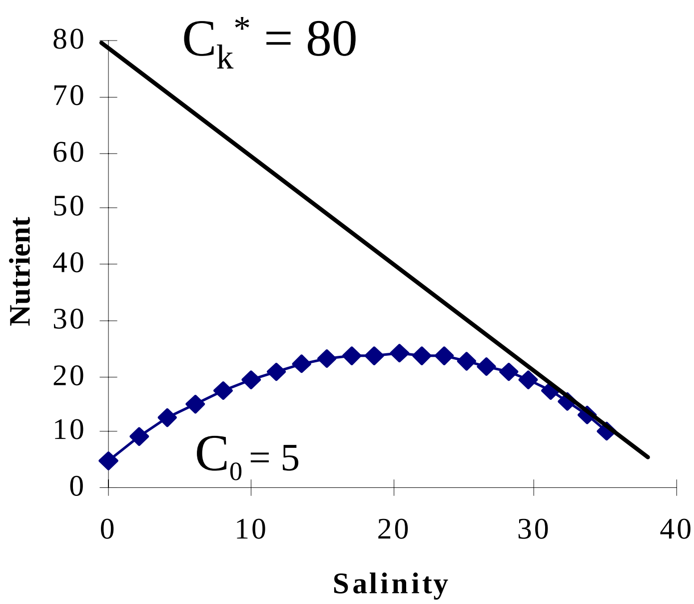


Figure IV.8 Example of nutrient parameter cross plot.

The flux of nutrient species from the catchment may be added to or reduced by biogeochemical processes operating in the estuary. The loss or gain of an element within the estuary can be determined from the difference between the flux from the river (QCk_0) and the flux across the marine isoconcentration surface.

$$L = QCk_0 - QCk^*$$

$$L = Q(Ck_0 - Ck^*)$$

Where Ck_0 is the steady state concentration of the component k in the river.

Thus the determination of a steady state mass balance using this model requires accurate definition salinity - property relationships within the estuary. Sampling is carried out on the basis of salinity gradients within an estuary rather than sampling at fixed stations.

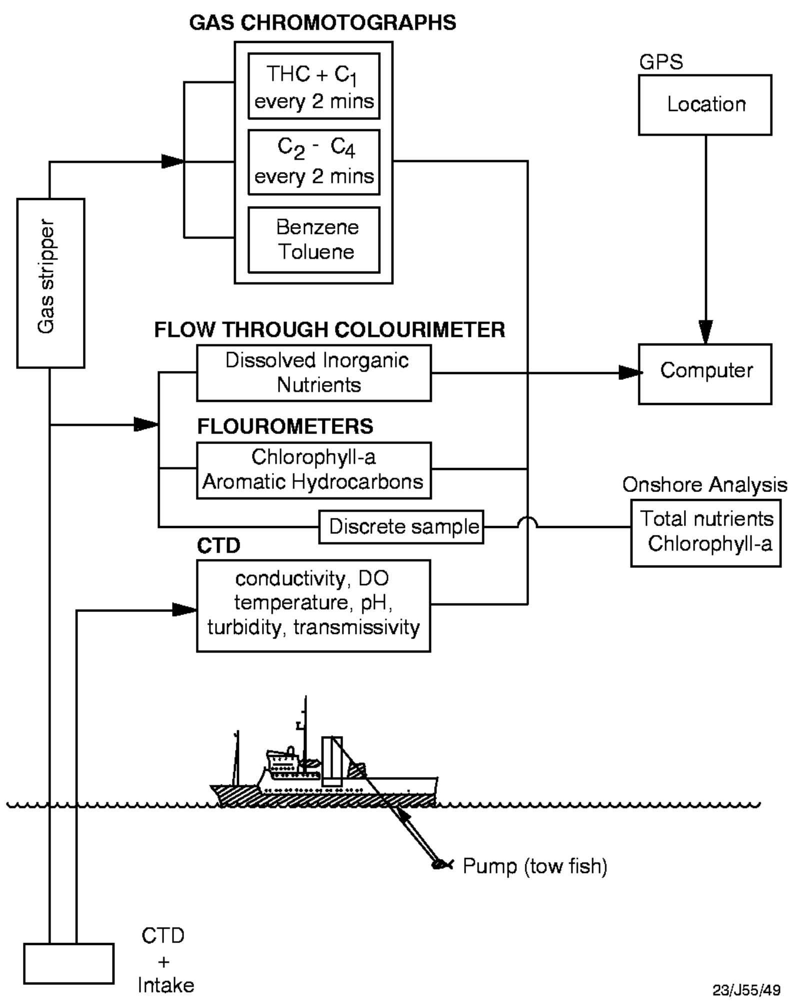


Figure IV.9 Flow diagram of the continuous sampling and analysis of seawater undertaken in this investigation.

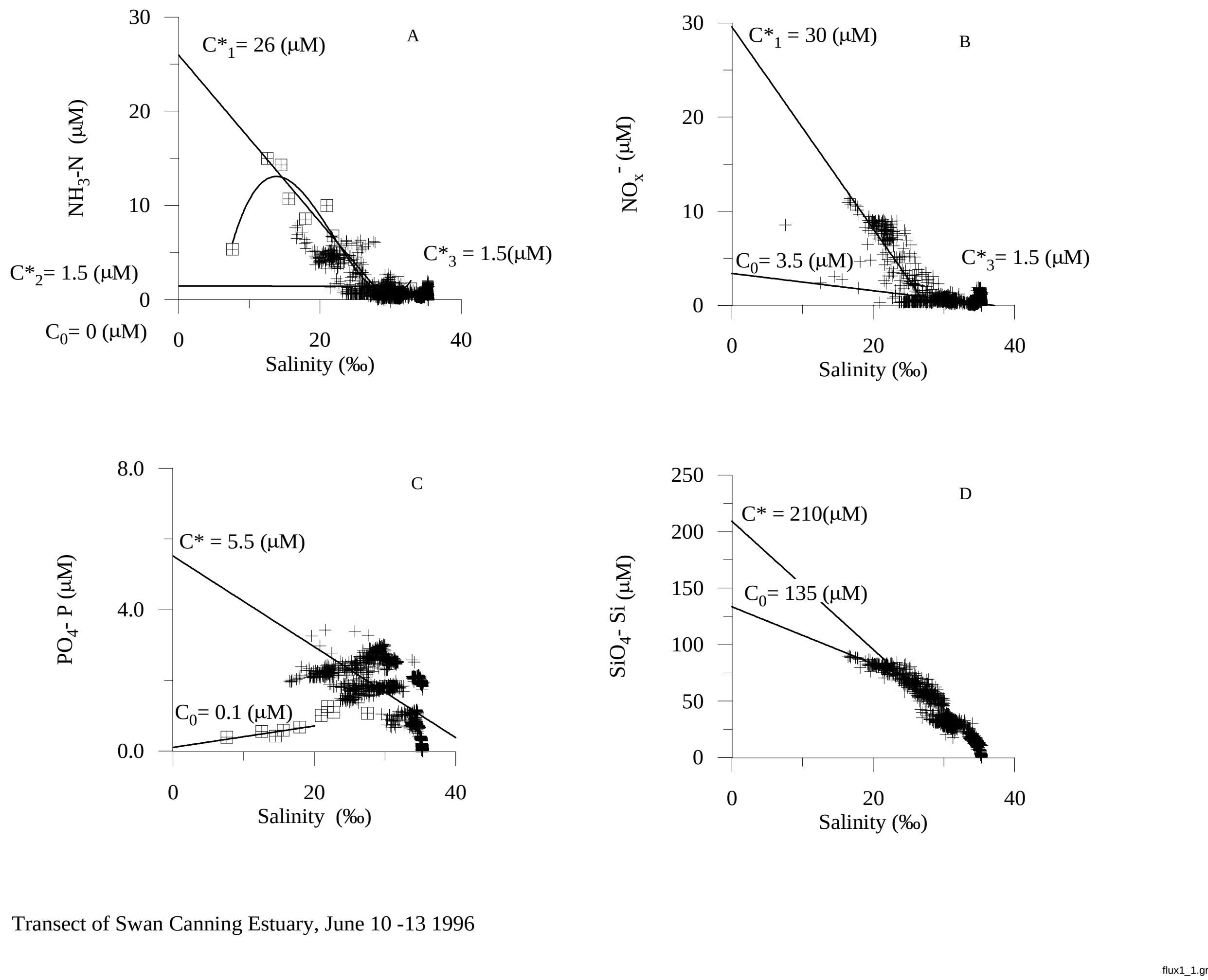


Figure IV.10 Survey 1 - Cross plot of salinity and dissolved inorganic nutrients - low flows, Swan Canning estuary.

Most investigators have used discrete sampling to define salinity-property relationships, sampling at each 1‰ change in salinity (Eyre and Twigg 1997, Kaul and Froelich 1984). We have used a system of real time continuous sampling as this provides higher resolution, the ability to identify small point sources and to modify sampling design in response to observed variations in water parameters.

<u>Continuous Geochemical Tracers (CGT)</u>

The study utilised continuous measurements of water properties in two surveys undertaken by AGSO and MAFRI. Continuous data collected in our surveys included:

- Temperature, salinity, dissolved oxygen (DO) and turbidity using a Yeo-Kal CTD.
- Automated measurement of dissolved nitrite (NO_2-N), nitrate (NO_3-N), ammonia (NH_3-N), orthophosphate (PO_4-P) and silicate (SiO_4-Si) in surface waters by flow through colorimetric methods and chlorophyll *a* by flow through fluorometry.

Vertical profiles of the water column were undertaken using a CTD probe at a number of locations within the estuary and nutrient samples were collected at 1m from the surface and 1m from the bottom. Samples were also collected for TN, TP, alkalinity and total suspended solids.

A flow diagram of the continuous sampling and analysis of seawater undertaken in this investigation is shown in Figure IV.9.

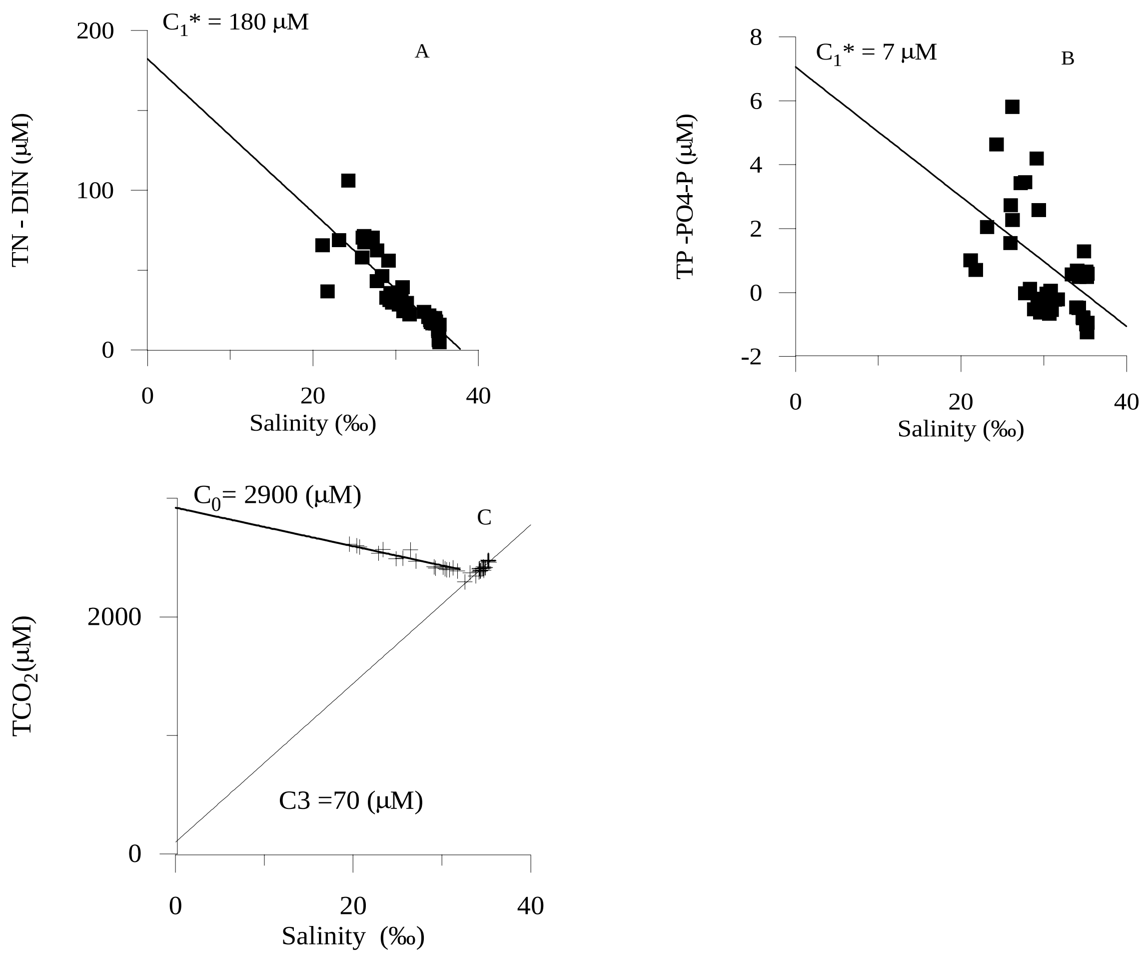


Figure IV.11 Survey 1 - Salinity parameter cross plot for early winter - low flows, Swan Canning estuary. (Concentration of N and P in Fremantle Harbour/Blackwall Reach were ignored in the surface layer model. This water enters the estuary as a bottom layer and was not part of the surface layer considered in this analysis. There was an unresolved discrepancy between NO_X measured by AGSO/MAFRI and bottle samples analysed by WRC.)

<u>Winter - Low flow</u>

Salinity/parameter cross plots for low flow conditions at the beginning of winter are shown in Figures IV.10 and IV.11. Nutrient fluxes estimated for the upper and lower estuary (surface layer) are shown in Table IV.2.

Table IV.2 Nutrient budget - low winter flows, Swan Canning estuary.

Upper Estuary CATCHMENT FLUX (F_Q)							
		TCO_2	NH_3-N	NO_X-N	TN-DIN	PO_4-P	TP-PO_4-P
Q	10^6 m^3day	n/d	0.05	0.05	0.05	0.05	0.05
Co	μM	n/d	5	30	180	0.1	7
Flux	moles/day	n/d	250	1500	9000	5	350
Upper Estuary ⇒ Lower Estuary FLUX (F25)							
Q	10^6 m^3day	0.05	0.05	0.05	0.05	0.05	0.05
C*	μM	2900	26	30	180	2	7
Flux	moles/day	145 000	1300	1500	9000	100	350
Lower Estuary - Flux (F35)							
Q	10^6 m^3day	0.05	0.05	0.05	0.05	0.05	0.05
C*	μM	70	1.5	1.5	180	5.5	7
Flux	moles/day	3500	75	75	9000	275	350

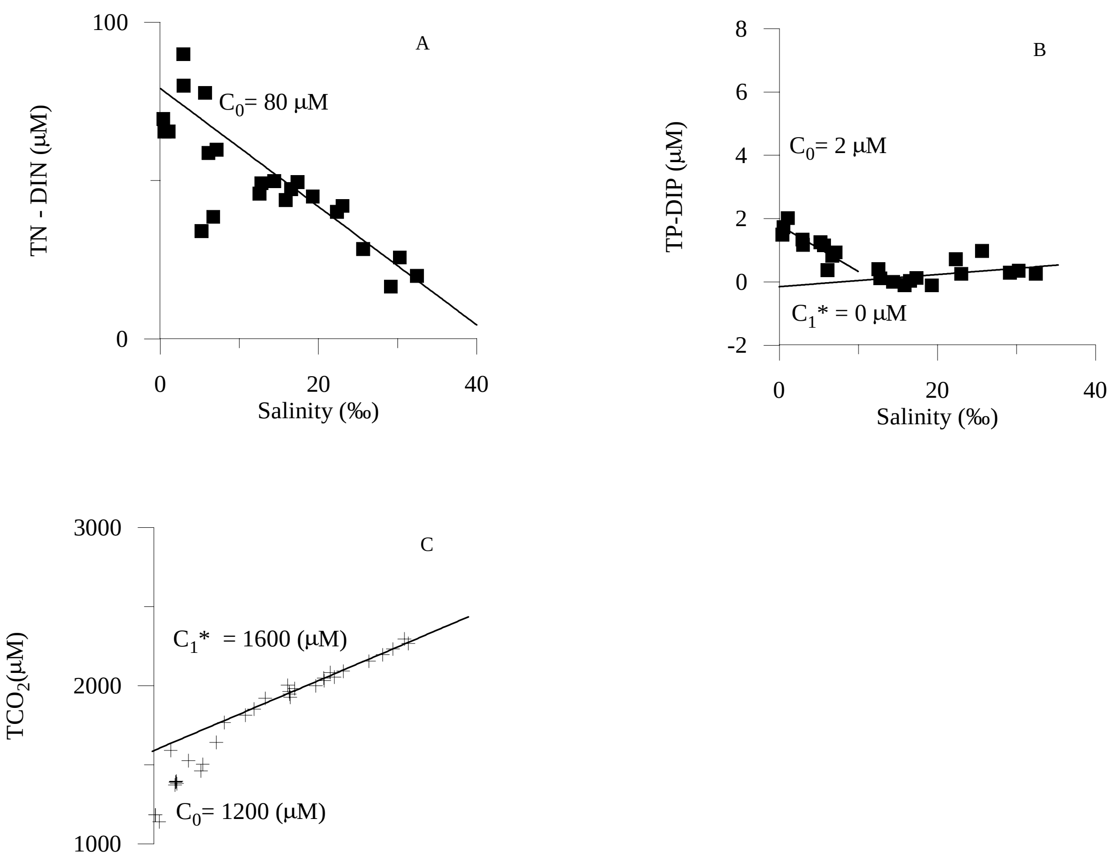
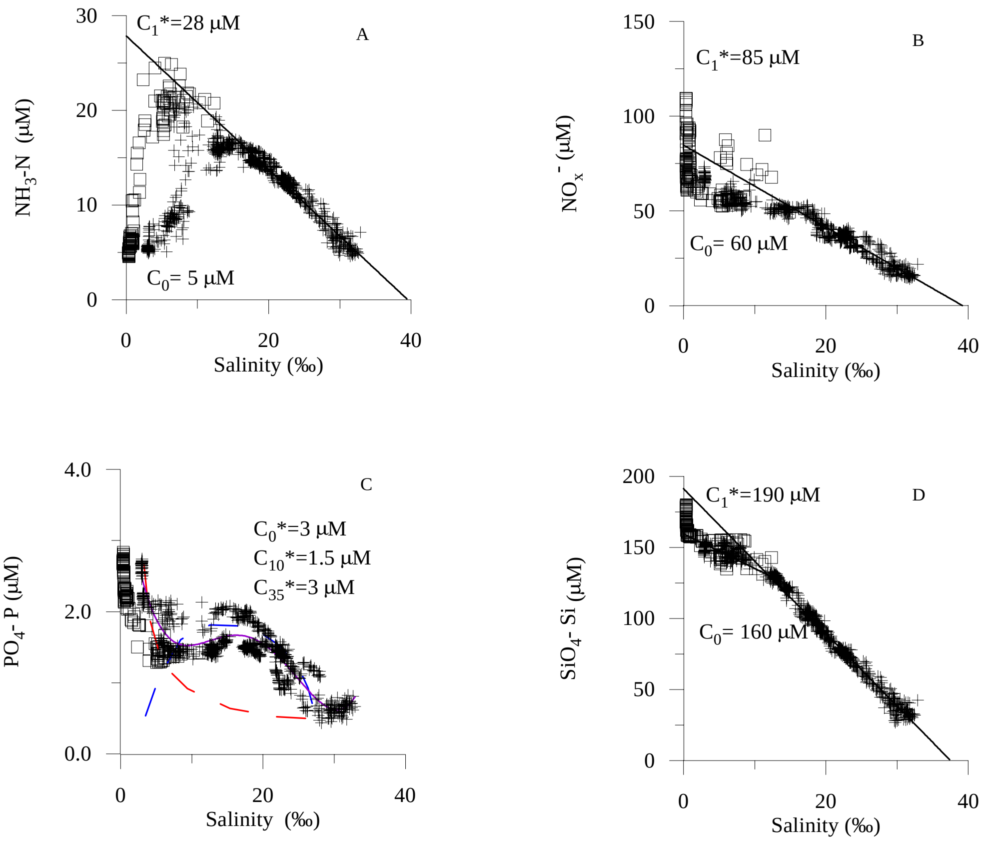


Figure IV.12 Salinity parameter cross plot - high winter flows, Swan Canning estuary.



Transect of Swan Canning Estuary, July 15 1996

flux3_1.grf

Figure IV.13 Salinity parameter cross plots - winter high flow, Swan Canning estuary. (+ = Swan Estuary, = Canning. Discharge data unavailable for the Canning so this estuary was not included. Discharge from the Canning River is likely to be a minor component of the water budget since the catchment is small and contains a relatively large reservoir.)

Table IV.3 Nutrient budget - high winter flows, Swan Canning estuary.

Catchment Flux (F_Q)								
		TCO_2	NH_3-N	NO_2-N	NO_X-N	TN-DIN	PO_4-P	TP-DIP
Q	10^6 m^3day	15	15	15	15	15	15	15
Co	µM	1200	5	1	75	70	3	4
Flux	moles/day	18x10^6	75 000	15 000	1 125 000	1 050 000	45 000	30 000
Flux to Marine Waters								
Q	10^6 m^3day	15	15	15	15	15	15	15
C*	µM	1600	30	3.5	81	80	3	0
Flux	moles/day	24 x 10^6	450 000	52 500	1 215 000	1 200 000	45 000	0

Winter - High flow

This survey was undertaken only in the lower estuary as high river flows prevented access to the upper estuary/Swan River. Salinity/parameter cross plots for low flow conditions at the beginning of winter are shown in Figure IV.12 and IV.13. Nutrient fluxes estimated for the upper and lower estuary (surface layer) are shown in Table IV.3. This budget refers to the surface layer of the lower estuary only.

Historical Nutrient Data

The detailed surveys undertaken by AGSO represent only a part of the seasonal pattern of nutrient cycling within the estuary; specifically, low flows at the onset of the winter rains and high flows following major rainfall. To broaden the perspective on nutrient cycling we have analysed the water quality data collected by WRC in 1995/96. The seasonal data, combined with salinity/nutrient cross plots utilised in the previous section, were then used to infer the major processes that control nutrient concentrations and dynamics in the estuary for each season.

We identified three key periods based on the water quality data set and our detailed studies reported above:

1. Winter - high flows, high rates of nutrient input but short residence times, cold temperatures and low light which make conditions less favourable for significant nutrient uptake within the estuary;
2. Spring - a period when runoff from the catchment is still delivering nutrients to the estuary but the longer residence times, warmer temperature and greater light provide suitable condition for uptake of nutrients by phytoplankton.
3. Late Summer - low or zero flows result in the intrusion of marine waters, stratification of the estuary and a build up of nutrient concentration in bottom waters.

Discharge

The catchment discharge for each survey

Table IV.4) was estimated by averaging discharge from the Avon River over the freshwater replacement time for the estuary (Kaul and Froelich 1984), and assuming that the Avon River contributed about 80% of runoff.

Table IV.4 Estimate average flow for each date, Swan Canning estuary.

Date	Average Flow Rate (m^3 day^{-1})
High Flow - Winter 18 July 1995	13×10^6
Moderate Flow - Spring 24 October 1995	1.7×10^6
Low Flow - Summer 22 January 1996	-1.75×10^6

There was no recorded runoff from the Avon River in April and May 1996. During this period of the hydrologic year, the estuary behaves as a salt wedge estuary and there is a net flow of marine waters into the estuary. We have estimated the net volume of intruding marine water from the volume of the estuary and the observed change in average salinity. Examination of the salinity record for the Armstrong Station shows that marine waters started to intrude into the estuary in early August 1995 and we have used this date to estimate an average daily rate of marine water intrusion.

Apparent Concentrations

The apparent concentrations used to estimate each flux were determined by extrapolation of the nutrient/salinity relationships to predict apparent end-member concentrations. Where this proved difficult because of the nature of the data, we have estimated of end-member composition (mainly seawater) from other sources.

High Flow - 18 July 1995

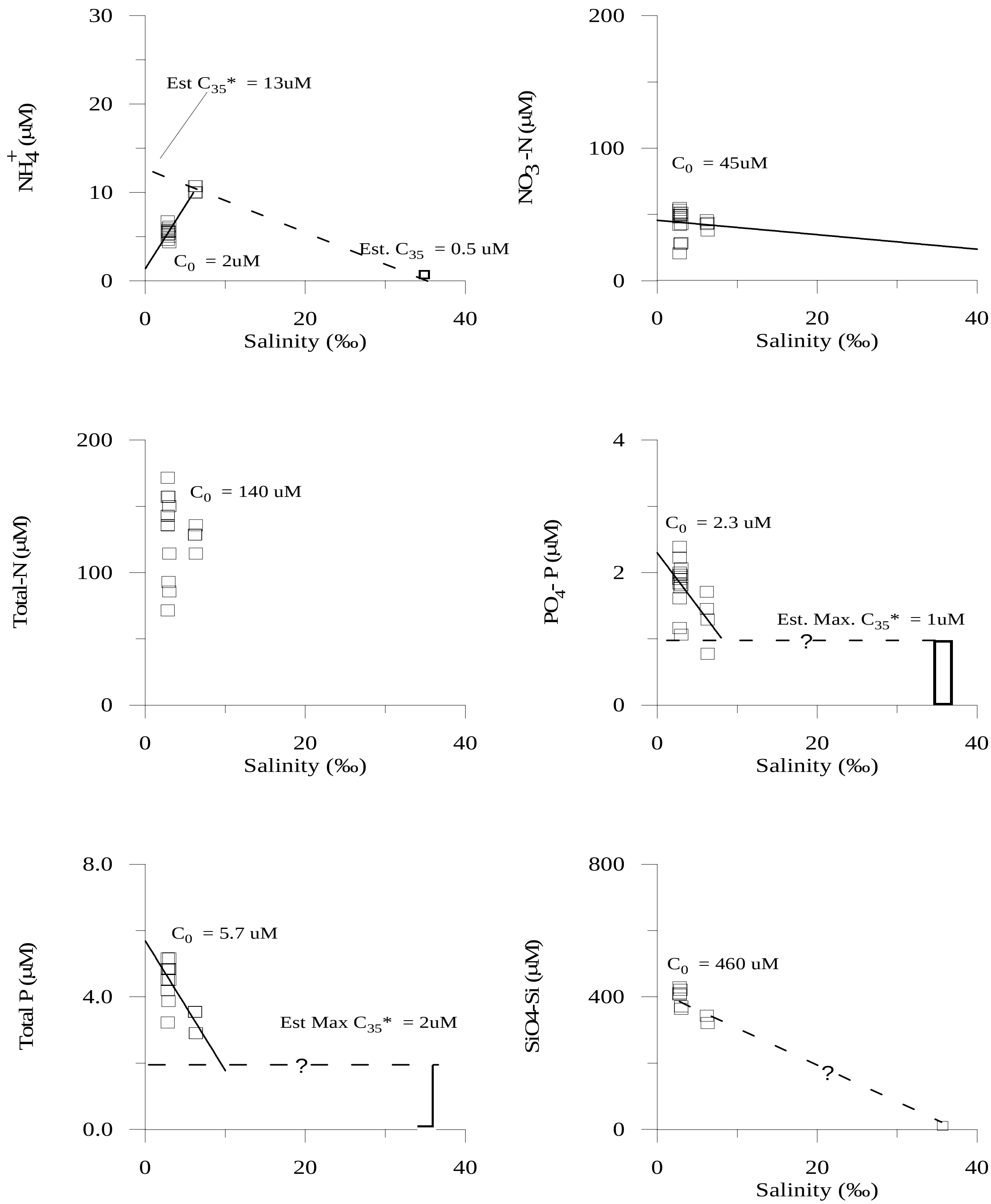
Winter rains started relatively early in 1995, the first significant runoff being recorded in late May and the first flood peak in June. By July 18, discharge had increased to about 13 x 10^6 m^3 day^{-1}, salinity had dropped to an estuary-wide average of about 5‰, and water temperature to about 15°C.

Cross plots of water quality parameters against salinity were constructed from WRC data collected on 18-July-1995 (Figure IV.14). The WRC data are limited (for this purpose) but are consistent with the more detailed interpretation based on our survey undertaken in July 1996. The WRC data show that:

1. the NH_4^+-N/salinity relationship must be "concave up" indicating a net input of NH_4^+-N within the estuary;
2. there are high concentration of both nitrate and total N in catchment runoff; and
3. there is evidence of removal of DIN and Total P within the estuary.

Table IV.5 Estimated fluxes - winter 1995, Swan Canning estuary.
(The estuary is treated as a single box.)

	Catchment Flux	Internal Flux	Flux Out
	(moles/day)		
NH_4	26 000	143 000	169 000
NO_x-N	585 000	0	585 000
PO_4-P	29 900	-16 900	13 000
SiO_4	5 980 000	0	5 980 000
TN-DIN	1 170 000	0	1 170 000
TP - PO_4-P	44 200	0	44 200



Swan Estuary - concentration of nutrients in surface waters plotted against salinity.
WRC monitoring 18-07-95

950718xp.grf

Figure IV.14 Winter - nutrient/salinity cross plots (18 July 1995), Swan Canning estuary.

Spring - 24 October 1995

By October, discharge from the catchment had dropped to about 1.4 x 10^6 m^3 day^{-1}, or about one tenth of that in July. In addition, water temperatures had increased to an estuary-wide average of about 18°C and surface salinities were about 10‰.

Cross-plots of nutrient concentration versus salinity for 24 October 1995 are shown in Figure IV.15. Again, the data are fragmentary as they were only collected over a salinity range of about 0 -20‰, and do not include the marine end-member. We have included estimates of marine concentration of each nutrient to facilitate interpretation.

It can be seen from Figure IV.15 that

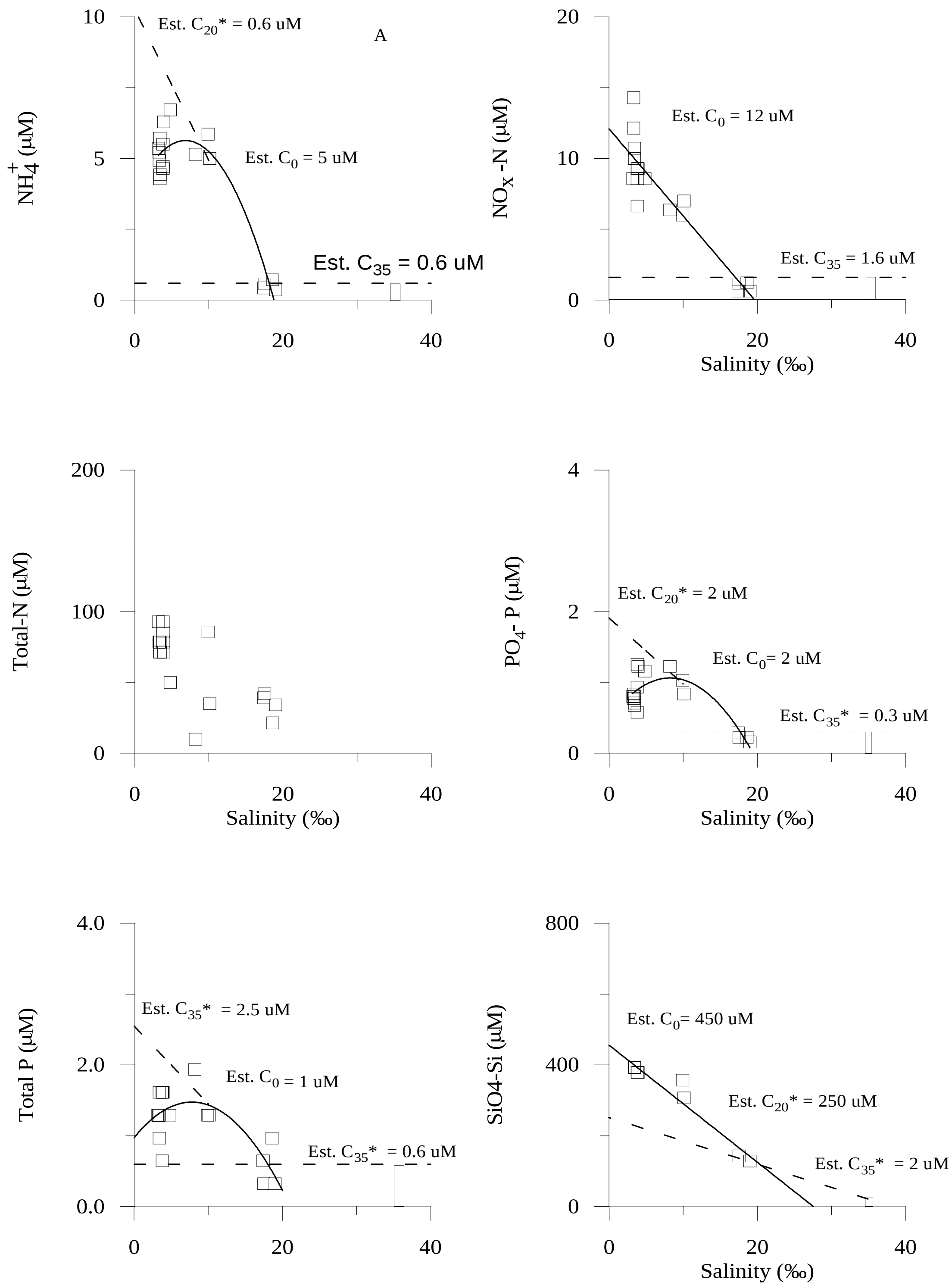
1. the concentration of NH_4^+-N shows a complex relationship with salinity. The relationship is apparently "concave up" at low salinities indicating internal input of NH_4^+-N in the upper estuary. However, the concentration of NH_4^+-N in the lower estuary is significantly less than expected from simple mixing with a marine end-member and this suggests removal of NH_4^+-N in the lower estuary (Perth and Melville Waters);
2. the relationship between NO_x-N and salinity also indicates N removal at higher salinities, but there is no evidence for any internal input in the upper estuary;
3. total nutrient data suggest conservative behaviour of TN;
4. P data are limited but suggest internal input of both DIP and TP at low salinities and removal at higher salinities (lower estuary); and
5. silicate data are limited but there is some evidence of removal.

Table IV.6 Estimated fluxes – spring 1995, Swan Canning estuary.

	Catchment Flux	Internal Input (Upper)	Internal Removal (Lower)	Net Flux
	moles/day			
NH_4	8500	8500	-15 980	1020
NO_x-N	20 400	0	-17 680	2720
PO_4-P	1190	2210	-2890	510
SiO_4	765 000	765 000		1 530 000
TN-DIN	1 170 000	0		
TP - PO_4-P	13 000	0		

Summer - 22 January 1996

By January, discharge from the catchment had dropped to near zero and there was a net inflow of marine water into the estuary. Water temperatures had increased to an estuary-wide average of about 27°C and salinities to an average of about 28‰.



Swan Estuary - concentration of nutrients plotted against salinity.WRC monitoring 24-10-95

951024xp.grf

Figure IV.15 Spring - nutrient/salinity cross plots, Swan Canning estuary.

Cross plots of water quality parameters against salinity for 22 January 1996 are shown in Figure IV.16. Again, the data are fragmentary and were only collected over a salinity range of about 20 -35‰, which does not include the freshwater end-member.

It can be seen from Figure IV.16 that

1. the concentration of N species in the estuary is low;
2. the concentrations of both NH_4^+ and NO_X show a mid-estuary maximum indicative of internal input, and
3. DON is the dominant N species and behaves conservatively.

The silicate data from this survey was uninterpretable. However, other surveys undertaken during summer showed a slightly "concave up" mixing curve between seawater with a low silicate concentration mixing with high silicate freshwater indicative of a net internal input within the estuary.

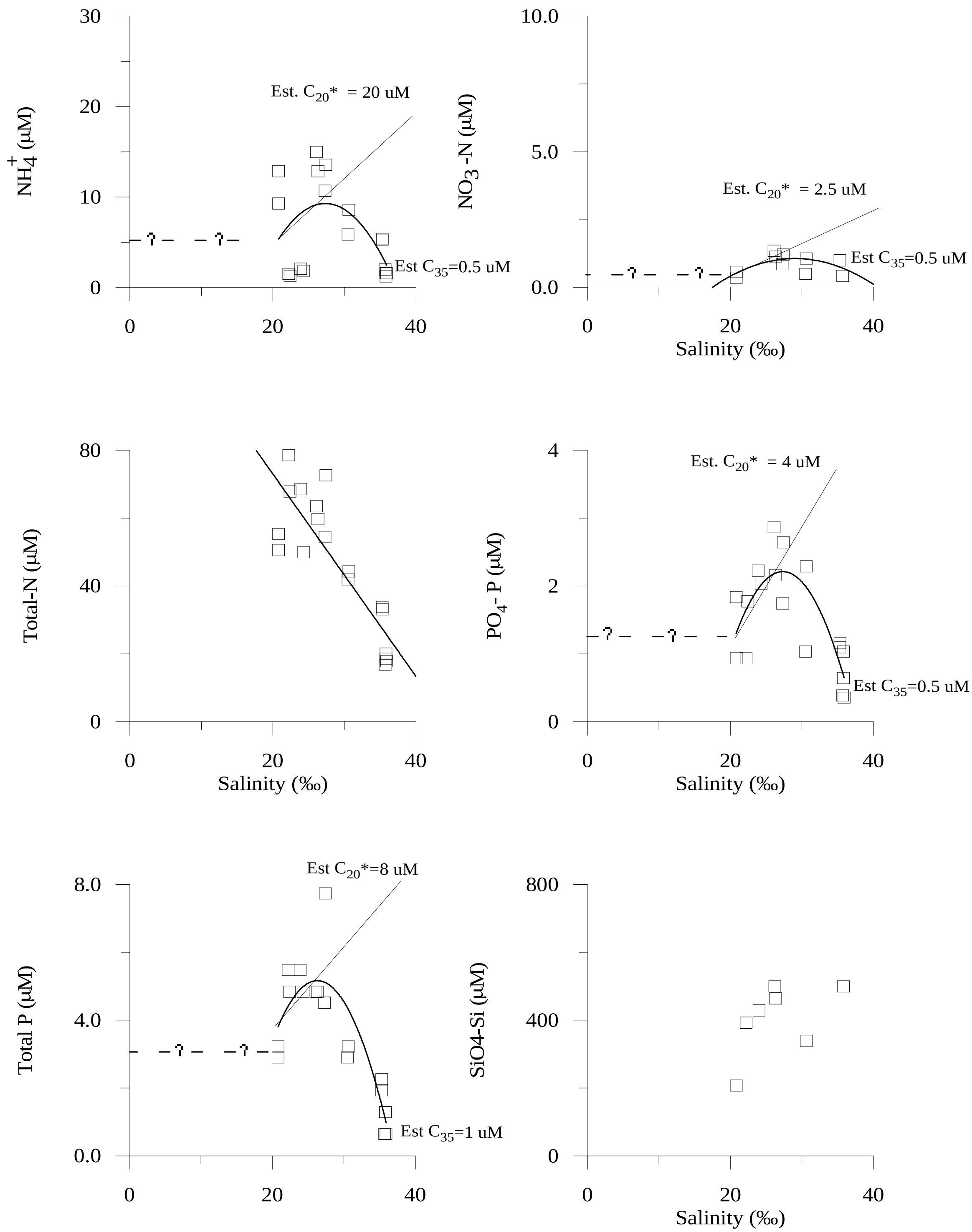
We cannot fully interpret these data without measurements of salinity and nutrient concentrations in the upper reaches of the estuary between Success Hill and Ellen Brook (salinities less 20‰). Extrapolation of the data suggest that there is a net removal of nutrients in the area but this needs confirmation from additional sampling.

Estimated fluxes are presented in Table IV.7. External nutrient fluxes during summer are small in comparison with other seasons and we suggest that any phytoplankton blooms occurring in summer must derive the bulk of their nutrients from internal recycling.

Table IV.7 Estimated fluxes – summer, Swan Canning estuary.

	Flux towards catchment	Net Removal Upper Est.	Internal Flux Lower Est.	Flux from marine water into estuary
	moles/d			
NH_4-N	9000	-27 000	35 100	900
NO_x-N	900	-3600	3600	900
PO_4-P	1800	-5400	6300	900
SiO_4	0	0	-18 000	18 000
TN-DIN	36 000	0	0	36 000
TP - PO_4-P	3600	-3600	6300	900

<u>Uncertainties</u>
Uncertainties in the flux estimates are difficult to determine as many of the interpretations have been made by extrapolation from typical marine concentrations. Uncertainties in estimates of apparent concentrations probably approach 50% in some cases. While this uncertainty may appear large it should be remembered that many of the net fluxes under investigation differ by an order of magnitude or more. In addition, the shape of the nutrient salinity relationship provides valuable qualitative information on the net balance of nutrient uptake and recycling within the estuary.



Swan Estuary - concentration of nutrients plotted against salinity.WRC monitoring 20-2-96

960220xp.grf

Figure IV.16 Summer - nutrient/salinity cross plots, Swan Canning estuary.

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APPENDIX V Workshop Report

1. Welcome

Participants were welcomed to CSIRO Land and Water by the Division Chief, Dr Graham Harris, who highlighted both the Australian scientific initiatives being taken in the coastal arena, especially by CSIRO, and the continuing efforts of CSIRO Land and Water to further its international activities and profile.

Participants were introduced and working documents were distributed.

2. Introduction and Background

2.1 LOICZ Core Project

An outline of LOICZ goals and approaches was presented by Dr Chris Crossland, who stressed the importance of the Workshop outcomes (see Terms of Reference, Appendix VII) to the continuing development of understanding of global change in the coastal zone within the International Geosphere-Biosphere Programme (IGBP). Key elements of the Project place emphasis on determining horizontal material fluxes at localities and sites, scaling site information to the regional and global dimensions by typological (coastal classification) methods, and linking flux information to the human dimension. The pivotal nature of derived biogeochemical budgets within LOICZ was highlighted, and the links to other elements of the Project (river catchments, typology, human dimension) were briefly described.

2.2 Overview of Workshop

The Workshop leader, Prof Steve Smith, described the purpose of the Workshop and the approach and progress of LOICZ in developing biogeochemical budgets. The LOICZ protocol for biogeochemical budgets in estuaries is one approach, among a number that could be used. However, this provides a common approach that allows comparisons between global sites, particularly for regions with limited data describing the relevant estuarine and coastal parameters. The development of a global statement by end 2002, depends on LOICZ accessing and using available data for the budgets and, by use of a series of typologies, to extrapolate regional sites information to a picture of the world's coastal zone. In addition, several relatively data-rich sites and regions should be explored in detail to extend the overall first-order assessment to areas of second and third order budgets which allow further assessment of forcing functions and system responses.

Realising a global assessment by LOICZ will require more than 100 site budgets. In addition to the budget developments, the work is delivering new tools for assessment of system function, such as relationships between salt and tidal exchange times.

The program for the Workshop and for report preparation was outlined for guidance of participants.

2.3 Systems Comparisons - Australian Estuaries and Lakes

The status and special nature of the Australian estuaries and embayments was outlined by Dr Graham Harris, noting the level of scientific understanding of the pressures and system processes. A range of features and issues were considered including their "marine" nature, their limited freshwater inputs, the often relatively long water residency times, and their sensitivity to land use and hydrology. Generally they are shallow and have strong macrophyte habitats, and they do not remain in "a steady state" for long due to seasonal variations in rainfall. Consideration was given to the implications of these factors on the types of models required to represent system processes.

The plenary presentation is contained in Appendix I.

2.4 Australian Coastal Zone Factoids

An overview of the physical and geographical diversity of Australasian estuaries was provided by Dr Bradley Opdyke, with contributions from Dr Brad Eyre about the classification systems being developed to describe the diversity of systems (see Appendix II). It is apparent that for many localities there is a large amount of data available, on the web and in other public domain sites, which is relevant to biogeochemical budget development. These data need to be compiled and integrated into systems models for use in management and expanding our understanding of the region's coastal dynamics.

3. Presentation of Australasian Biogeochemical Budgets

The budgets contributed to the Workshop covered a range of regional areas and climatic conditions.

The contributed budgets for the systems were briefly considered by participants, including an overview of the system settings, data availability, approaches being taken to build the biogeochemical budgets, and the status and problems in development of estimates. Several existing budgets were reviewed in light of new information. Systems presentations included:

Modelled with LOICZ methods

Swan River, Western Australia	Dr Malcolm Robb & Linda Kalnejais
Cockburn Sound, Western Australia	Dr Tony Chiffings
Wilson's Inlet, Western Australia*	Dr Malcolm Robb
Hardy Inlet, Western Australia	Dr Malcolm Robb
Port Phillip Bay, Victoria	Drs John Parslow & Graham Skyring
Gippsland Lakes, Victoria	Drs Phillip Ford & Ian Webster
Derwent River, Tasmania	Dr John Parslow
Lake Illawarra, New South Wales	Dr John Morrison & Kathie MIller
Hawkesbury-Nepean system, New South Wales	Gary Bickford
Queensland and northern NSW systems (11)	Dr Brad Eyre
Central Great Barrier Reef, Queensland*	Dr Miles Furnas
Fly River, Papua New Guinea	Dr Bradley Opdyke
Hauraki Gulf, New Zealand	Dr John Zeldis

* currently listed at the LOICZ biogeochemical budget models website (www.nioz.nl./loicz/); additional data are being incorporated

Modelled with non-LOICZ methods ***(see Appendix IV)***

Richmond estuary, New South Wales	Lester McKee and Dr Brad Eyre
Swan Canning system, Western Australia	Dr David Fredericks

Throughout the presentations, there were discussions on a range of issues, both generic and specifically related to the individual systems, including:

- how to assess estuaries regarding the freshwater inputs and evaporation processes,
- the implications for evaluation of data by aggregation to annual averages and/or by seasonal sets,
- dealing with stratified systems, and
- the use of multi-box model approaches.

Water residency and retention times, and the impact of seasonal and episodic events on budget modelling were considered, especially in relation to the "big picture" classification of the region, previously presented in the "Factoid" session.

4. Budgets Development

Break-out groups worked interactively on the development of these and additional site budgets, supplemented with methodological and site/issues-based tutorials and discussions. Estimates for sites and evolution of assessment approaches were made, often incorporating more detailed spatial and temporal boxes and data in the models. Budget refinements were made in light of outcomes from individual and group discussions of issues emerging from additional plenary sessions.

5. Additional Plenary Sessions and Discussions

Two additional plenary sessions were developed from early discussion about various site budgets:

- Sensitivity and Analysis of Models
  Drs John Parslow and Ian Webster delivered a consideration of the errors and assessment of sensitivity for the biogeochemical budgets derived through the LOICZ approach, and the implications of episodic river flows on the methodology. Appendix III contains the derived discussion paper addressing these issues.

- Preliminary latitudinal comparisons and patterns in the nutrient budget data and systems assessment

DR BRAD EYRE PROVIDED AN OUTLINE OF INITIAL LATITUDINAL PATTERNS EMERGING FROM THE WORKSHOP RESULTS. THESE OBSERVATIONS ARE CONTAINED IN PART IN THE WORKSHOP OVERVIEW (SECTION 1) AND APPENDIX II, AND A WIDER CONSIDERATION IS BEING PREPARED FOR PUBLICATION IN GLOBAL LITERATURE.

6. Outcomes and Wrap-up

Completed budgets for all systems were developed to a final stage of completion; some required additions to text descriptions and a check on data sources before contribution. Participants provided copies of their complete estimates for inclusion in the Workshop Report and for lodgement on the LOICZ website.
A number of additional sites were identified for which data is available and which may potentially yield budgets. Participants committed to making contact with other researchers for data and either to carry out or to encourage others to make further site evaluations for contribution to LOICZ. It became apparent through the Workshop that there are a number of localities in Australia for which there are relevant time-series data (more than a decade), often extending across management responses to nutrient enrichment input to the systems. For example, in the Hawksbury-Nepean system sewage treatment plants have been installed or modified to higher treatment technology. Also, in the Swan River system land use and channelisation management has been put in place. These areas could provide a valuable assessment of nutrient load impacts and remediation effects on the estuarine processes, fitting the LOICZ objective of gaining an understanding of links between the human dimension and processes influencing biogeochemical cycles.

The timetable for delivery of final budgets and publication of the Workshop Report was established: all contributions for the Report to be provided by 30 November 1998 with additional budgets to be contributed by 15 November 1998. The latter to be included in the Report and in a CD ROM containing the full regional information from this and a subsequent workshop in Mexico.

The participants joined with LOICZ in expressing thanks to the local organiser, Dr Bradley Opdyke, and Dr Graham Harris and staff of CSIRO Land and Water for support and for hosting the Workshop. The financial support for the Workshop by CSIRO Land and Water was gratefully acknowledged by LOICZ.

APPENDIX VI Workshop Participants and Contributors

Gary Bickford
Sydney Water, Level 17
PO Box 453,
Sydney South, NSW 2000
AUSTRALIA
Telephone 61 2 9350 5382
Fax 61 2 9350 5929
Email gary.bickford@sydneywater.com.au

Gregg Brunskill[1]
Australian Institute of Marine Science
PMB No.3, MSO
Townsville, Qld 4810
AUSTRALIA
Telephone 61 7 4753 4111
Fax 61 7 4772 5852
Email G_Brunskill@aims.gov.au

Tony Chiffings
Australian Water Technologies
PO Box 73
West Ryde, NSW 2114
AUSTRALIA
Telephone 61 2 9334 0711
Fax 61 2 9334 0792
Email tony_chiffings@awtensight.nsw.gov.au

Chris Crossland
LOICZ International Project Office
Netherlands Institute for Sea Research
PO Box 59, 1790 AB
Den Burg, Texel
THE NETHERLANDS
Telephone 31-222-369404
Fax 31-222-369430
Email loicz@nioz.nl

Peter Davies[1]
Centre for Coastal Management
Southern Cross University
PO Box 157
Lismore, NSW 2480
AUSTRALIA

Vilma Dupra[1]
Department of Oceanography
University of Hawaii
Honolulu, Hawaii 96822
USA
Telephone 1 808 956 2354
Fax 1 808 956 7112
Email vdupra@soest.hawaii.edu

Bradley Eyre
Centre for Coastal Management
Southern Cross University
PO Box 157
Lismore, NSW 2480
AUSTRALIA
Telephone 61 2 6620 3773
Fax 61 2 6621 2669
Email beyre@pophost.scu.edu.au

Phillip Ford
CSIRO Land and Water
GPO Box 1666
Canberra, ACT 2601
AUSTRALIA
Telephone 61 2 6246 5559
Fax 61 2 6246 5560
Email Phillip.Ford@cbr.clw.csiro.au

David Fredericks
Australian Geological Survey Organisation
Canberra, ACT 2601
AUSTRALIA
Telephone 61 2 6249 9434
Fax 61 2 6249 9589
Email dfrederi@agso.gov.au

Miles Furnas
Australian Institute of Marine Science
PMB No.3, MSO
Townsville, Qld 4810
AUSTRALIA
Telephone 61 7 4753 4323
Fax 61 7 4772 5852
Email mfurnas@ aims.gov.au

Graham Harris
CSIRO Land and Water
GPO Box 1666
Canberra, ACT 2601
AUSTRALIA
Telephone 61 2 6246 5620
Fax 61 2 62465564
Email Graham.Harris@clw.csiro.au

David Heggie
Australian Geological Survey Organisation
Canberra, ACT 2601
AUSTRALIA
Telephone 61 2 6249 9589
Fax 61 2 6249 9980
Email dheggie@agso.gov.au

Linda Kalnejais
Water and Rivers Commission WA
Level 2 Hyatt Building
3 Plain Street
East Perth, WA 6001
AUSTRALIA
Telephone 61 8 9278 0525
Fax 61 8 9278 0586
Email linda.kalnejais@wrc.wa.gov.au

A. Longmore[1]
Marine and Freshwater Resources Institute,
PO Box 114
Queenscliff, Vic.3225
AUSTRALIA

Lester McKee
Centre for Coastal Management
Southern Cross University
PO Box 157
Lismore, NSW 2480
AUSTRALIA
Telephone 61 2 6620 3493
Fax 61 2 6621 2669
Email lmckee@scu.edu.au

Kathryn McMahon[1]
Water and Rivers Commission WA
Level 2 Hyatt Building
3 Plain Street
East Perth, WA 6001
AUSTRALIA

Cathee Miller
Environment Research Institute
University of Wollongong
Wollongong, NSW 2522
AUSTRALIA
Telephone 61 2 4285 4495
Fax 61 2 9330 6035
Email cmiller@spider.connectivity.net.au

John Morrison
Environment Research Institute
University of Wollongong
Wollongong, NSW 2522
AUSTRALIA
Telephone 61 2 4221 4134
Fax 61 2 4221 4665
Email john_morrison@uow.edu.au

Bradley Opdyke
Department of Geology
Australian National University
Canberra, ACT 0200
AUSTRALIA
Telephone 61 2 6249 4205
Fax 61 2 6249 5544
Email bno@geology.anu.edu.au

John Parslow
CSIRO Marine Research
GPO Box 1538
Hobart, Tas. 7001
AUSTRALIA
Telephone 61 3 6232 5202
Fax 61 3 6232 5000
Email John.Parslow@marine.csiro.au

Peter Pepperell[1]
Centre for Coastal Management
Southern Cross University
P.O. Box 157
Lismore, NSW 2480
AUSTRALIA

Malcolm Robb
Water and Rivers Commission WA
Level 2 Hyatt Building
3 Plain Street
East Perth, WA 6001
AUSTRALIA
Telephone 61 8 9278 0526
Fax 81 8 9278 0586
Email malcolm.robb@wrc.wa.gov.au

Graham Skyring
Skyring Environment Enterprises
40 Atherton Street
Downer, ACT 2602
AUSTRALIA
Telephone 61 2 6241 1036
Fax 61 2 6241 9662
Email see@ozemail.com.au

Stephen V. Smith
Department of Oceanography
University of Hawaii
Honolulu, Hawaii 96822
USA
Telephone 1 808 956 8693
Fax 1 808 956 7112
Email svsmith@soest.hawaii.edu

Atsushi Suzuki[1]
Australian Institute of Marine Science
PMB No.3, MSO,
Townsville, Qld 4810
AUSTRALIA
Telephone 61 7 4753 4111
Fax 61 7 4772 5852
Email asuzuki@pearl.aims.gov.au

Robert Wasson
Department of Geogaphy
Australian National University
Canberra, ACT 0200
AUSTRALIA
Telephone 61 2 6249 2475
Fax 61 2 6249 3770
Email robert.wasson@anu.edu.au

Ian Webster
CSIRO Land and Water
GPO Box 1666
Canberra, ACT 2601
AUSTRALIA
Telephone 61 2 6246 5581
Fax 61 2 6246 5560
Email Ian.Webster@cbr.clw.csiro.au

John Zeldis
10 Kyle Street
Riccarton, Christchurch
NEW ZEALAND
Telephone 64 6 3348 8987
Email j.zeldis@niwa.cri.nz

[1]Did not attend workshop but contributed to budget preparation.

APPENDIX VII Workshop Agenda

LOICZ Workshop on Australasian Estuaries
CSIRO Land & Water,
Canberra, Australia
12-14 October, 1998

Agenda

Monday, October 12th

0830 Visitors meet in University House Lobby
0900 Welcome from Graham Harris, CSIRO in Pye Bldg, Seminar Room
0910 Chris Crossland, Comments from the LOICZ IPO
0920 Steve Smith, Short overview of workshop
0935 Graham Harris, Overview of system comparisons
1005 Brad Opdyke, Factoids about Australasian coastal zone
1100 John Morrison and Cathy Miller, Lake Illawarra, NSW
1140 Malcolm Robb and Linda Kalnejais, Wilson Inlet, WA
1220 John Parslow, Graham Skyring, Port Phillip Bay, Victoria
1300 LUNCH (on site)
1400 Ian Webster and Phillip Ford, Gippsland Lakes, Victoria
1440 Tony Chiffings, Cockburn Sound, WA
1600 Bradley Eyre and Lester McKee, Northern NSW estuaries
1700 Discussion

Tuesday, October 13th

0900 CSIRO Meeting Room, Comments from Steve Smith
0910 Dave Fredericks and Dave Heggie, Swan River, WA
0950 Bradley Opdyke, Fly River, Papua New Guinea
1100 John Zeldis, Hauraki Gulf, New Zealand
1140 Miles Furnas, GBR Lagoon, Queensland
1220 Plenary discussion: Where from here?
1300 LUNCH (on site)
1400 Breakout discussion and writing groups
1530 Continue discussion and writing groups
1700 Plenary discussion

Wednesday, October 14th

0900 CSIRO Meeting Room, Comments from Steve Smith
0910 Continue discussion, writing groups
1100 Continue discussion, writing groups
1200 Plenary discussion - Status of budgets
1300 LUNCH
1400 Plenary discussion - Regional and latitudinal comparisons
1530 Adjourn

APPENDIX VIII Terms of Reference for Workshop

TERMS OF REFERENCE
LOICZ WORKSHOP ON AUSTRALASIAN ESTUARINE SYSTEMS
CSIRO Land & Water, Canberra, Australia
12-14 October 1998

Primary Goals:
To work with researchers dealing with Australasian estuarine systems, in order to extract budgetary information from as many systems as feasible from existing data. The Australasian systems (including Australia, Papua New Guinea and New Zealand) span a climatic regime ranging from cool (arid and wet) temperate to both wet and dry tropics; they vary from relatively little to high degree of perturbation from human activities; and many Australasian estuarine systems are very intensively studied. These systems thus provide potential proxies for many areas with relatively little information. The potential latitudinal gradient for these systems is almost 40 degrees. Information to budget many of these systems is available, and there is an active scientific community of researchers working on these lagoons. This workshop will complement an earlier, very successful workshop held in Ensenada, Mexico, in June 1997, a second Mexican workshop in January 1999 (Merida, Mexico), and a South American workshop to be held in November 1999 (Bahia Blanca, Argentina) by the analysis of data from another well-studied region which overlaps and extends beyond many of the climatic, hydrological and latitudinal characteristics of Mexico and Central America.

Anticipated Products:
1. Develop budgets for as many systems as feasible during the workshop.

2. Examine other additional data, brought by the researchers or provided in advance, to scope out how many additional systems can be budgeted over an additional few (~ 2) months.

3. Contribution of these additional sites to two or three papers to be published in the refereed scientific literature: (a) In combination with expected output from the Mexican studies, a paper comparing the biogeochemical functioning of estuaries in arid regions. (b) In combination with expected output from the Mexican and South American workshops and available data from the U.S. and perhaps Canada, a paper on latitudinal gradients in estuarine biogeochemical functioning. (c) A regional paper on comparison of estuarine biogeochemical function over the hydrological and climatic gradients of Australasia.

Participation:
The number of participants will be limited to less than 20 persons, to allow the active involvement of all participants. Nominees include:
- Technical secretariat support (Chris Crossland);
- LOICZ SSC Members (Steve Smith);
- Graham Harris (CSIRO) and Bradley Opdyke (ANU) as local organisers;

- Researchers from Australia and New Zealand research institutes.

Workplan:
Participants will be expected to come prepared to participate in discussions on coastal budgets. Preparation should include reading the LOICZ Biogeochemical Modelling Guidelines (Gordon *et al.*, 1996), the Mexican Lagoons Workshop Report (Smith *et al*., 1997), examination of the tutorials presented on the LOICZ Modelling web page (http://data.ecology.su.se/MNODE/) and arriving with spreadsheets containing available budgeting information from "their sites."

Each participant should arrive with a draft of at least one water/salt/nutrient budget set, generally following the LOICZ procedures. It would be helpful if participants also brought a draft writeup (1-3 text pages, + site map), in electronic form plus "budget boxes" (hand-drawn for the boxes is okay; these will be drafted according to a common format). Examples can be found in the "Mexican Lagoons" workshop report. For the sake of consistency, please express rates as annual and in molar (rather than mass) units.

Background Documents (for reference, to meet LOICZ initiatives):

1. Gordon, D.C., Boudreau P.R., Mann K.H., Ong J.-E., Silvert W., Smith S.V., Wattayakorn G., Wulff F., and Yanagi T. 1996. LOICZ Biogeochemical Modelling Guidelines. LOICZ Reports and Studies 5, 96 pp.
2. Smith S. V., Ibarra-Obando S., Boudreau P.R., and Camacho-Ibar V.F. 1997. Comparison of Carbon, Nitrogen and Phosphorus Fluxes in Mexican Coastal Lagoons. LOICZ Reports and Studies 10, 84 pp.
3. LOICZ Modelling web page, for everyone with www access: (http://data.ecology.su.se/MNODE/)

APPENDIX IX Glossary of Abbreviations

NH_4	Ammonium
NO_3	Nitrate
DIN	Dissolved inorganic nitrogen
DON	Dissolved organic nitrogen
DIP	Dissolved inorganic phosphorus
DOP	Dissolved organic phosphorus
PTN	Particulate total nitrogen
PTP	Particulate total phosphorus
ON	Organic nitrogen
OP	Organic phosphorus
TN	Total nitrogen
TP	Total phosphorus
DOC	Dissolved organic carbon
DIC	Dissolved inorganic carbon
POC	Particulate organic carbon
OC	Organic carbon
SiO_4	Silicate
nfix	Nitrogen fixation
ndenit	Denitrification
p	Primary production
r	Respiration
TDN	Total dissolved nitrogen
TDP	Total dissolved phosphorus
CTD	Conductivity Temperature Depth