GOVERNMENT OF WESTERN AUSTRALIA

WATER QUALITY IN WILSON INLET 1995-1997

A report on the monitoring data collected between 1995 and 1997

WATER RESOURCE TECHNICAL SERIES

WATER AND RIVERS COMMISSION REPORT WRT 14
1999

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Cover Photograph: Aerial view of Wilson Inlet looking east. [Photo by Simon Neville]

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Linda Kalnejais and Malcolm Robb

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Summary

The Waters and Rivers Commission has routinely monitored water quality in Wilson Inlet between 1995 and 1997. Seven sites have been sampled at weekly intervals for nutrients (dissolved inorganic and total nitrogen and phosphorus species and silica) and physical parameters (temperature, salinity, colour and dissolved oxygen). Phytoplankton cell counts have also been determined. This report presents the data that was collected and discusses the results.

The hydrodynamics of the Inlet in 1995 to 1997 can be characterised into four states, modified from those proposed by Marshall (1993). Each state is described below. Other important events associated with each state are also described:

The Closed Dry State (typically January/February – May/June): The Inlet water column was well mixed with low levels of nutrients and oxygenated waters. There was little river flow and the salinity increased over this period due to evaporation.

The Closed Wet State (typically June/July – August): The start of the winter rainfall changed the state of the Inlet slightly. Inflows from the rivers formed freshwater plumes on the surface, but these were destroyed by wind mixing, resulting in an Inlet-wide reduction in salinity. Freshwater plumes in the eastern basin were only able to form after there was sufficient runoff and moved downstream as a buoyant overflow. The runoff carried in significant levels of inorganic oxidised nitrogen (NOx), total nitrogen (TN), silica and colour.

The Open Wet State (typically August – October/November): After the mechanical opening of the sand bar, marine water intruded into the Inlet within 4 days. The intrusion moved into the western basin, forming a saline bottom layer up to 1 metre in depth. Up to two weeks later the saltwater penetrated into the eastern basin, where the saline layer reached a thickness of 20 cm. Due to continued freshwater discharge, the Inlet behaved as a salt wedge estuary over this time. Wind mixing and convective cooling were generally insufficient to completely destroy the salinity stratification. Only strong and continuous winds (moderate gales) were able to completely break up the stratification. The well mixed water column did not persist for long after strong winds, as the stratification was re-established by successive tidal inflows bringing new marine water into the Inlet.

The oxygen levels in the Inlet showed strong dependence on the salinity structure. Deoxygenated waters were reported below the halocline several days after salinity stratification became established. Persistent stratification led to severe deoxygenation, and anoxic conditions were reported on several occasions.

High levels of ammonium were released from the sediments when anoxic conditions were reported. Elevated levels of filterable reactive phosphorus and silica were also often associated with anoxic conditions. The high levels of these species observed under anoxic condition indicate there was a change in the geochemical pathways under anoxic conditions, leading to increased nutrient release from the sediments. Sulphate reduction of organic matter in the sediments may have been an important process.

The Open Dry State (typically October/November – December/January): With the reduction in streamflow, the channel to the ocean shoaled and restricted tidal exchange. Marine water intrusions did, however, continue. The stratification was less severe than that reported in the ‘Open Wet state, and wind mixing was able to mix the water column frequently. The salinity of the Inlet increased over this period. The oxygen levels in the water column were influenced by the strength and persistence of the salinity stratification.

A phytoplankton bloom occurred in October and November in each year sampled (1995 through to 1997). Dinoflagellates and diatoms were the dominant species involved.

The nutrient levels declined over this period, reaching baseline levels that remained steady until the start of

the winter rains and the ‘Closed Dry’ State.

The four hydrodynamic states observed were distinct in each year of this study, however the time and duration of each state was variable from year to year. Important factors controlling the hydrodynamic state of the Inlet were the rainfall, wind patterns, atmospheric pressure and tidal elevations.

Two distinct nutrient sources were evident in the Inlet: an external nutrient source originating from the catchment and an internal nutrient source derived from the sediments. The concentrations of NOx, ammonium and FRP fell in the water column more than can be accounted for by dilution and flushing alone. The Inlet was thus retaining a quantity of these nutrients.

The phytoplankton bloom reported in each year would have captured most of the dissolved nutrients within the water column. Following the collapse of the phytoplankton bloom, dissolved nutrients were briefly detected in the water column as the cells decomposed. However after this, the nutrient level in the water column remained low through the ‘Open Dry’ and ‘Closed Dry’ states. Any nutrients released to the water column over this period were probably taken up by *Ruppia* seagrass and its epiphytes and benthic algae.

The data from this study was compared with water quality data collected over 1982/83 (Lukatelich *et al*, 1984) to assess if any changes in water quality could be identified. The mean summer nutrient levels of the 1995-1997 data were comparable or lower than those reported in the 1982/83 data. Higher winter means of NOx in the surface waters and ammonium in the bottom waters were reported in 1995-1997. The mean levels of organic nitrogen and phosphorus were higher in 1982/83 than in 1995 to 1997 data.

The water quality in Wilson Inlet on 1995 to 1997 was generally good. For over half the year nutrient levels were close to detection limits, there was little phytoplankton activity and the water was clear. Over winter, however the water column nutrient levels rose due to catchment input and internal sediment sources. Biological uptake captured much of the dissolved inorganic nitrogen and phosphorus species, and the Inlet acts as a significant nutrient sink. The biomass of the Inlet (*Ruppia*, epiphytes, phtyoplankton) was thus acting to keep the nutrient levels in the water column low. The Inlet does, however, show symptoms of potentially more severe nutrient enrichment problems. Should the anoxic events increase in spatial extent or duration, then the internal store of organic matter in the sediments could be released in greater quantities than has been currently observed. This may lead to a shift in species composition from *Ruppia* seagrass and to favour less ecologically useful macroalgae and more intense phytoplankton blooms.

Continued water quality monitoring is recommended to act as a surveillance system and to provide further understanding of the important processes influencing nutrient cycling.

To ensure that the Inlet water quality does not degrade, different measures need to be taken. Catchment based programmes are essential to reduce the nutrients and particulates that flow into the Inlet. Fencing creeklines to limit stock access, revegetating creeklines, converting septic systems to sewerage and applying only the minimum required amount of fertiliser, will all reduce the amounts of nutrients that reach the Inlet.

1. Introduction

Wilson Inlet is a coastal lagoon on the south coast of Western Australia, adjacent to the township of Denmark. The Inlet runs parallel to the coast and is 14 km long with a maximum width of 4 km and maximum depth of 5 metres. The total area of the Inlet is 48 km^2. A catchment of 2379 km^2 drains to the Inlet. The catchment extends northward for 48 km and is a maximum of 60 km wide (Figure 1). The town of Mount Barker lies on the north -eastern catchment boundary. Four major tributaries flow through the Wilson Inlet catchment, the Denmark, Hay, Sleeman Rivers and Cuppup Creek. Over 70 % of the catchments of the Hay, Cuppup and Sleeman have been cleared for agriculture, while much of the Denmark River catchment remains forested.

At the mouth of the Inlet there is a sandbar. The streamflow from winter rains raises the waterlevel in the Inlet. Once the Inlet level exceeds 1.02 m above mean sea level, the sandbar is artificially breached to prevent flooding of the low lying areas around the Inlet. In most years the sandbar has generally been opened in July or August (Hodgkin and Clarke, 1988). The channel that is formed remains open for several months, during which time there is exchange between the Inlet and the ocean. With decreased streamflow in summer the channel shoals and eventually closes, so that marine water exchange is no longer possible and the waterbody is isolated from the ocean.

Over the past decade there have been reports of algal blooms and excessive growth of the seagrass *Ruppia megacarpa* within the Inlet (Lukatelich et al, 1982, Hodgkin and Clark, 1988). These observations of increased aquatic plant growth may be a symptom of nutrient enrichment, or eutrophication of the Inlet. If the nutrient enrichment of the Inlet is allowed to progress unchecked, over the course of several years (maybe decades) the Inlet water quality may degrade, and algal blooms and other nuisance plant growth could become more prevalent. In order to intervene effectively against the eutrophication of Wilson Inlet, it is important to understand the sources and processes that are contributing nutrients to the Inlet and the processes controlling the cycling and movement of the nutrients within the Inlet.

The Wilson Inlet Management Authority together with the Water and Rivers Commission commenced catchment and Inlet water quality monitoring in a response to the need for information on the processes operating in the Inlet and its catchment. The regular monitoring started in 1995, and has continued to the present day. This report presents the data that was collected between 1995 and 1997 and discusses the results.

1.1 Historical water quality data and previous studies

The water quality within Wilson Inlet has been the subject of several studies over short periods of time. Spencer (1952) published data collected between 1945 and 1950 as part of hydrological investigations in south-western Australia. Chlorinity, temperature, oxygen, pH, phosphate and nitrate levels were measured at 8 stations at approximately yearly intervals.

Salinity, temperature and oxygen data were collected during 1971 and 1972 at 8 sites (Lenanton, 1974) and presented in a report written in order to address the issues of changing the position of the bar opening.

In the early 1980's there was considerable concern in the local community about the apparently excessive growth of the seagrass *Ruppia megacarpa* Mason. Consequently the Department of Conservation and Environment commissioned a report on the nutrient status of Wilson Inlet. Lukatelich, Schofield and McComb (1984 and 1986) calculated nutrient loadings to the Inlet and made estimates of the Inlet biomass for both 1982 and 1984. The summary of their findings for 1982 was presented in Lukatelich *et al* (1987).

A considerable data set was collected for both study years. The data included nutrient concentrations in tributary inflows, sediment nutrient concentrations, macrophyte nutrient content and biomass. The water quality data collected in the Inlet consisted of 3 primary sampling sites at which salinity, dissolved

oxygen, temperature, nutrients and chlorophyll *a* were collected from the surface and bottom at 6 weekly intervals during 1982-1983. In 1984-1985 two intensive sampling surveys were carried out at 16 sites.

In 1982, a dry year, it was calculated that there was a net retention of phosphorus (60% of total input stored), while there was a net export of nitrogen from the Inlet (115% of total inputs exported). In 1984, which had rainfall close to the long-term average, it was calculated that there was a net retention of both phosphorus (49% of total input) and nitrogen (63% of total input).

Lukatelich, Schoefield and McComb (1984 and 1986) showed that the nutrient store in the *Ruppia* greatly exceeded the riverine loading in 1982 and was of the same magnitude as the riverine loadings in 1984. The water quality in the Inlet was generally good, but it was concluded that the Inlet was showing symptoms of eutrophication, as evidenced by the rapid growth of the *Ruppia*. Although *Ruppia* is a symptom of eutrophication, it is also the mechanism by which the nutrient levels in the water column are kept low.

Hodgkin and Clark (1988) published a report on Wilson Inlet, as part of a series on the estuaries of the south coast. This report discussed the catchment, geomorphology, vegetation, fauna and management of the Inlet. The water quality data described above was reviewed.

Water quality data were collected by several different groups between 1987 and 1994. The Murdoch University School of Biological and Environmental Sciences collected water quality data from 7 sites between 1987 and 1989 as part of research on the fish and aquatic wildlife of the Inlet. No report specifically on water quality has been prepared using this data.

In 1990 a data collection project was instigated by the Wilson Inlet Management Advisory Commitee and the Waterways Commission of Western Australia. The data collection for this project was carried out by staff of the Waterways Commission and Mr Owen Macintosh, a local fisherman and farmer. Up to ten sites were sampled during this programme between 1990 and 1993. Temperature and salinity data were collected at 0.5 metre intervals and nutrient samples were also collected.

The water quality data collected between 1990 and 1992 were the subject of a final year engineering research project at the Centre for Water Research. In this project Marshall (1993) analysed the salinity and temperature data and described the seasonal characteristics of the Inlet hydrodynamics. Four hydrodynamic states of the Inlet were distinguished.

The closed state: Due to the closure of the bar no seawater is able to penetrate into the Inlet. The density structure and circulation is influenced by wind mixing, river and groundwater flows, differential heating and cooling and evaporation.

The open winter state: Due to the high river inflows no seawater is able to penetrate the Inlet at any point of the tidal cycle. Net seawater propagation into the Inlet was estimated to occur when the total streamflow fell below 3 million cubic metres per day.

Salt wedge Condition: Seawater penetrates the Inlet mouth for some or all of the tidal cycle. The extent of saltwedge penetration depends on the streamflow and tidal range.

Open Summer State: As the streamflow reduces and the saltwedge penetrates into the Inlet the conditions become more saline. Strong sea breezes act to break down the vertical stratification of the Inlet, keeping it well mixed.

A specific and more detailed field study of the hydrodynamics of Wilson Inlet is given in Ranasinghe and Pattiaratchi (1996) and a numerical hydrodynamic model of the Inlet is described in Ranasinghe and Pattiaratchi (1998). The aim of the field study was to identify the dominant hydrodynamic processes of the estuary. The field study was conducted over 1994/95 with four intensive surveys, each lasting three days. The important processes in the hydrodynamic states proposed by Marshall were modified to account for the new data. The open winter state is described as a partially well mixed estuary state, with stratification and de-stratification occurring daily. The stratification during the day is due to solar radiation and streamflow and destratification during the night is due to mainly penetrative cooling and the occurrence of strong wind events (> 10 m/s). A saltwater intrusion of up to 2000 m was identified during the flooding tide, but was flushed out by the following ebb tide. No field data was collected to indicate the salt wedge condition, however, no field surveys were performed between

September and November (inclusive), the period when a salt wedge condition would be most likely to occur. Ranasinghe and Pattiaratchi (1996) remained sceptical as to the existence of a salt wedge condition, arguing that with the falling streamflow allowing marine water to penetrate, shoaling of the Inlet entrance would also occur, so limiting tidal exchange at the mouth and preventing a salt wedge condition.

The inflows and outflows to the Inlet were modelled by Boughton (1997). The simulations covered periods of many months and included both 'bar opened' and 'bar closed' states. This gave insights into the hydrodynamic processes which control the inflows and outflows to the Inlet, especially the flows between the Inlet and the ocean when the bar is opened. Boughton (1997) demonstrated that the flows are a function of the Inlet and ocean levels, and of the geometry of the shallows which extend about 2 km into the Inlet from the sandbar. A by-product of the simulation was the quantification of the daily tidal inflows and outflows for a period of 4.5 months from the bar opening in August to the end of December in each year.

1.2 Water and Rivers Commission weekly monitoring

In 1995 in a response to the need for more detailed, longer term information on the water quality in the Inlet, the Water and Rivers Commission commenced weekly monitoring of physical and chemical parameters and collected phytoplankton samples. This report discusses the data that was collected by this programme between 1995 and 1997.

The data from 1995 to 1996 has already been the subject of a report. The Water and Rivers Commission commissioned a review of the 1995 to 1996 data set to assess the phytoplankton ecology in the context of the water quality parameters of the Inlet (Thompson, 1997). Thompson (1997) found that nitrate enters the Inlet from riverine flow, while inorganic phosphorus and ammonium are not derived from riverine input but are associated with short term anoxic events in the water column. Short and moderate algal blooms occur 0-3 weeks after high levels of phosphorus occurred. The importance of microphytobenthos (benthic microalgae) in controlling nutrient release from the sediments was suggested due to their location and ability to oxygenate the sediments. The largest nutrient stores in the Inlet, namely the *Ruppia* and the sediment are both located within the euphotic zone due to the shallow nature of the Inlet. This fact combined with the occurrence of deoxygenated waters, which can block denitrification and lead to sediment ammonium release, indicate that the Inlet could be in a precarious balance (Thompson, 1997).

2. Methods

Water quality monitoring of Wilson Inlet has been undertaken on behalf of the Water and Rivers Commission at approximately weekly intervals since January 1995. On a routine sampling run data was collected from seven standard sites (Sites WI 30, WI 6, WI 7, WI 9, WI 12, WI 14 and WI 2 or WI 35) as shown on Figure 2. Both water samples for chemical analyses and physical data were collected at each site. Other sites were sampled opportunistically as required. The AMG co-ordinates of each site and the sampling regime for 1995 – 1997 are provided in Appendix A. For the purpose of this report the sites have sometimes been categorised into basins. The western basin is taken to be between Poddy Point and Pelican Point. This includes sites WI 6 and WI 35. The eastern basin is east of Pelican Point, and so includes sites WI 9, WI 12 and WI 14.

2.1 Chemical data

Samples were collected from the surface and bottom waters using a hose sampler. Samples for the analysis of dissolved chemical species were filtered immediately after collection with a 0.45 μm cellulose nitrate membrane. Further volumes of water were filtered through a Whatman GF/C filter (1 μm) for chlorophyll *a*, *b* and *c* and phaeophytin pigment analyses. The samples were stored in a refrigerator, at temperatures below 4ºC before being sent to Perth to the Australian Environmental Laboratories (formerly Analabs Ltd) for analysis. The principle species that were analysed over the three year period were ammonium, nitrate, total nitrogen, filterable reactive phosphorus, total phosphorus, silica and colour. Other species that have been analysed for shorter periods are total organic carbon, nitrite and selenium. The time periods over which each species was analysed is provided in Appendix A, together with the laboratory methods and detection limits.

2.2 Physical data

A Hydrolab H20 multiprobe (Hydrolab Corporation, Austin Texas, USA) was used at each site to collect temperature, salinity, specific conductivity and dissolved oxygen data at approximately 0.5 metre intervals through to the bottom of the water column. Details of the accuracy of the probe are provided in Appendix A. Secchi disk depths were also measured at each site using a 30 cm diameter secchi disk.

2.3 Biological data

Phytoplankton samples were collected from sites 6, 12 and 30. A rubber hose lowered through the water column was used to collect a water sample integrated over the depth of the site. The samples were fixed in Lugols Iodine preservative and sent to the Phytoplankton Ecology Unit of the Water and Rivers Commission for microscope identification. Identification was to the Class level for most organisms.

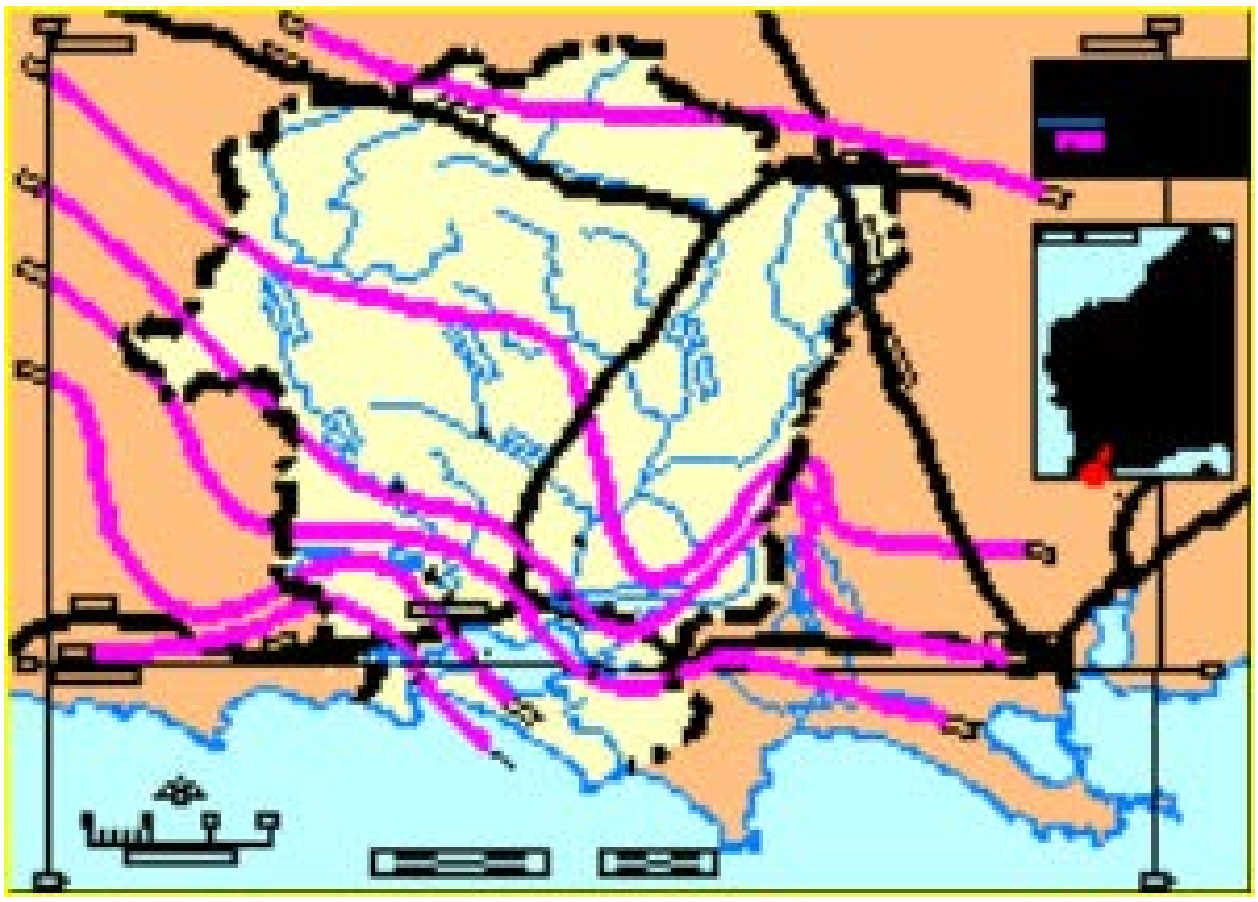


Figure 1. Wilson Inlet and catchment

Data

- The water quality data collected for Wilson Inlet has been verified and stored in the River and Estuaries Section's database EDICT.
- River flow data was verified and stored using HYDSYS software.
- Albany tidal data was supplied by the Coast and Marine Section of the Department of Transport.
- Albany wind speed and direction data was supplied by the Bureau of Meteorology.

Data analysis

Data analysis and graphic presentations of the data were completed using Statsoft's Statistica and Golden Software's Surfer software packages.

Figure 2. Wilson Inlet showing the locations of sampling sites

Depths are referenced from 0.2 m above AHD.

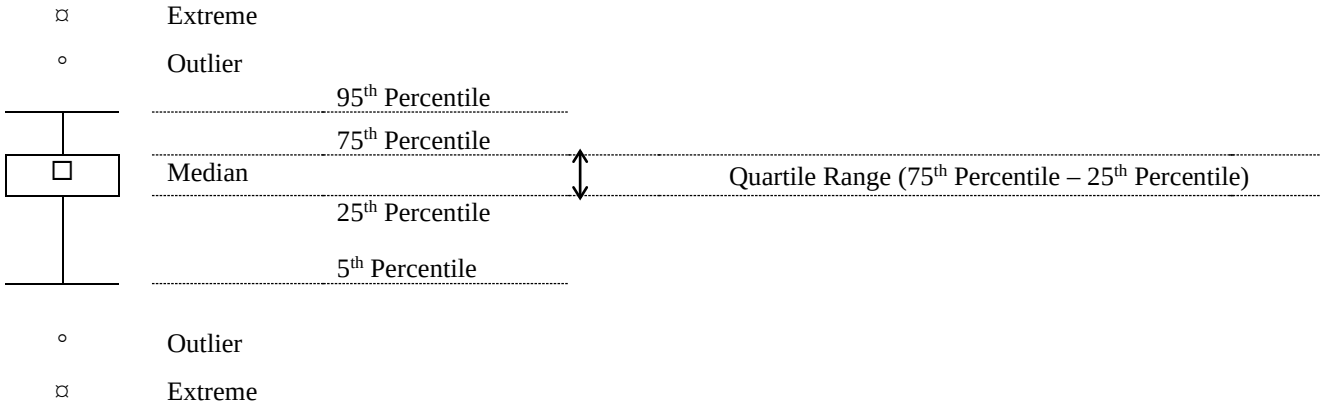
3. Results

From 1995 to 1997 a total of 138 sampling runs were conducted. The data for each major parameter are presented below. The data are presented in terms of both the seasonal variations and variations by site.

In Section 4 the data are analysed in more detail, with particular emphasis on determining the important processes influencing the water quality and long term health of the Inlet.

3.1 Seasonal variation

The data presented in this section are discussed in terms of categorised box and whisker plots. Box and whisker plots are used to summarise the distribution of data. This is done by plotting some of the key statistical values to give an indication of the values and spread of the data. Box and whisker plots indicate the central tendency of the data set, the variability around the central point (the box), and give an indication of the range of the data (the whiskers). The box and whisker plots used in this report use the median to indicate central tendency and percentiles to indicate the box and whisker portions of the graph. This is shown in the diagram below:



The plots used in this report also indicate values that are considerably different from the rest of the population. These points are plotted individually and are referred to as outliers and extremes. These are defined by the following:

A point is termed an outlier if:
data point value > 75th percentile + 1.5 × (quartile range)
or
data point value < 25th percentile - 1.5 × (quartile range)

A point is termed an extreme if:
data point value > 75th percentile + 3 × (quartile range)
or
data point value < 25th percentile - 3 × (quartile range)

3.1.1 Physical data

Temperature

The water temperature within the Inlet followed a pronounced seasonal pattern (Figure 3). The maximum temperature occurred in January/February, with a median of 21.7 ºC. The water temperature then declined, reaching its lowest monthly median value of 12 ºC in July. The temperature rose back to above 20 ºC by December. The variance in the temperature was low, with 50% of the data for each month (ie the quartile range) within 2 ºC. There was very little difference between sites as each site showed a similar temperature range.

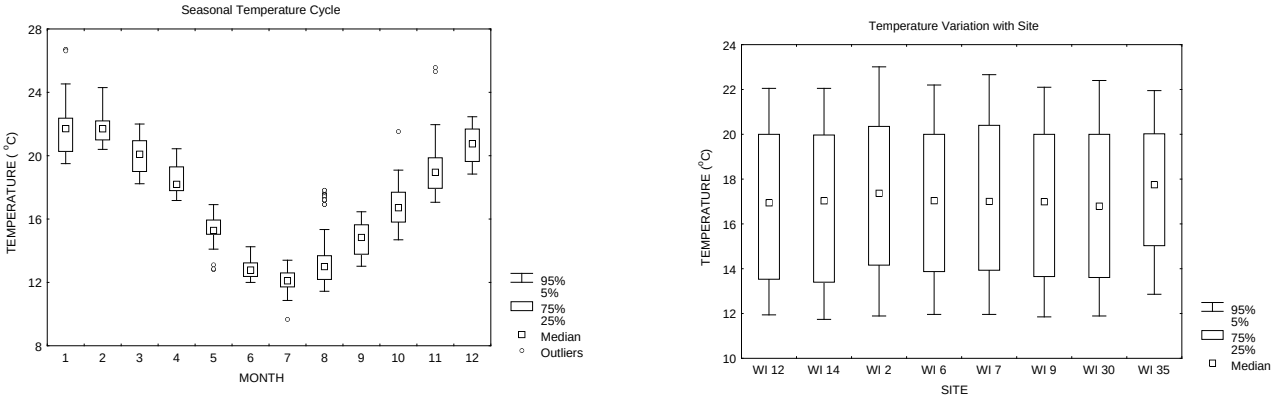


Figure 3. Seasonal temperature cycle and temperature variation between sites

Salinity

The Inlet salinity followed a distinct seasonal cycle (Figure 4). In September/October the overall Inlet salinity was at it lowest, with a monthly median value of around 16 ppt. The median salinity then rose by between 2 and 3 ppt/month until January. The salinity continued to rise, but at a slower rate until May/June, reaching a maximum monthly average salinity of around 27 ppt. From June to October the average salinity fell. Most of the water column in Wilson Inlet followed this seasonal cycle, as the quartile range was less than 5 ppt for all months other than July. From March to May the Inlet was extremely homogeneous, often with a difference of less than 2 ppt across the entire Inlet. However, there were a considerable number of times between August and February when the salinity at certain sites and depths did not behave the same as the rest of the population.

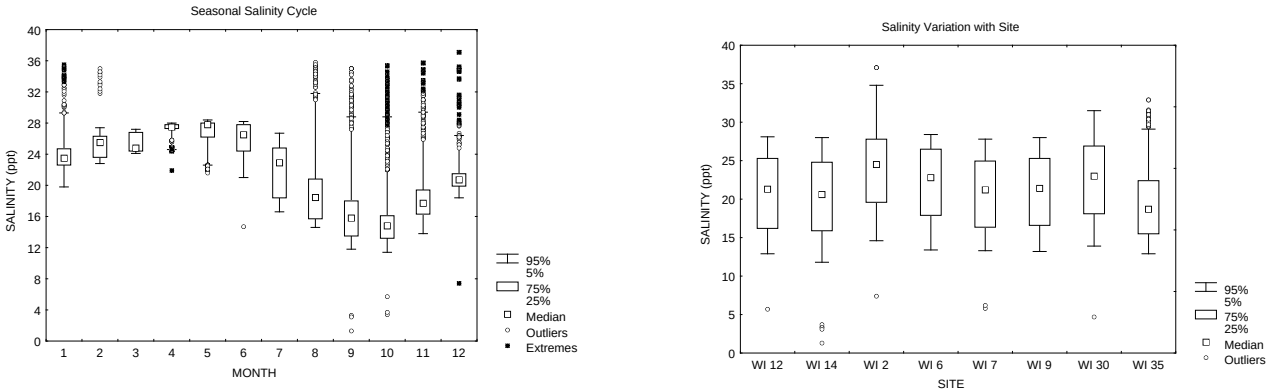


Figure 4. Seasonal salinity cycle and salinity variation between sites

Extreme salinity values occurred that were both significantly higher than the quartile population and significantly lower. The occurrence of these extreme values was dependent on the location of the site. Observations of salinity readings below 10 ppt only occurred at sites WI 7, WI 14 and WI 12 and only between late August and October. Low salinities occurred mainly in the surface waters, less than 0.5 m deep. Site WI 14 showed the highest number of low salinity readings.

From September to December salinities greater than 24 ppt, which were significantly higher than the monthly averages for these months were reported at all sites

other than WI 7. At WI 14, WI 12 and WI 9 the saline water occurred as a bottom most layer to a maximum depth of 20 cm and on only a few occasions per year. At sites WI 6 and WI 35 the higher salinity water was reported as a bottom layer to a depth of 1 metre, and was detected in most sampling runs conducted during this period.

Marine salinities (above 30 ppt) were observed from August to February at sites WI 2, WI 30, WI 6 and WI 35. Site WI 2 reported marine water throughout the entire water column on several occasions each year, while the other sites reported marine water as a bottom layer with a maximum depth of 1 metre at WI 30 and 0.5 metres at WI 6. It can therefore be seen that the extremes in salinity shown in Figure 4 illustrate the period when the Inlet is stratified due to the inflow of marine water, or less frequently due to the inflow of river water.

Dissolved oxygen

The median dissolved oxygen concentration, when expressed as percentage saturation, was constant throughout the year, varying between 92 and 100% saturation, and showed no seasonal pattern (Figure 5). In contrast the range of dissolved oxygen values expressed in mg/L varied greatly and showed a strong seasonal pattern. There was a median concentration of 7.1 mg/L in January/February and 9.5 mg/L in July (Figure 6). All samples were collected during daylight hours, with 80% of the samples collected between 9 am and 1 pm. Between March and July the variation in oxygen levels was lower than observed at other times of the year, with most observations close to the mean and no values falling below 60% saturated (approximately 5 mg/L) or rising above 140% saturated (approximately 10 mg/l). As with salinity, for the rest of the year more than half of the data population was within 10% of the mean value. However there were a significant number of sites and depths, which displayed values considerably above or below the median values.

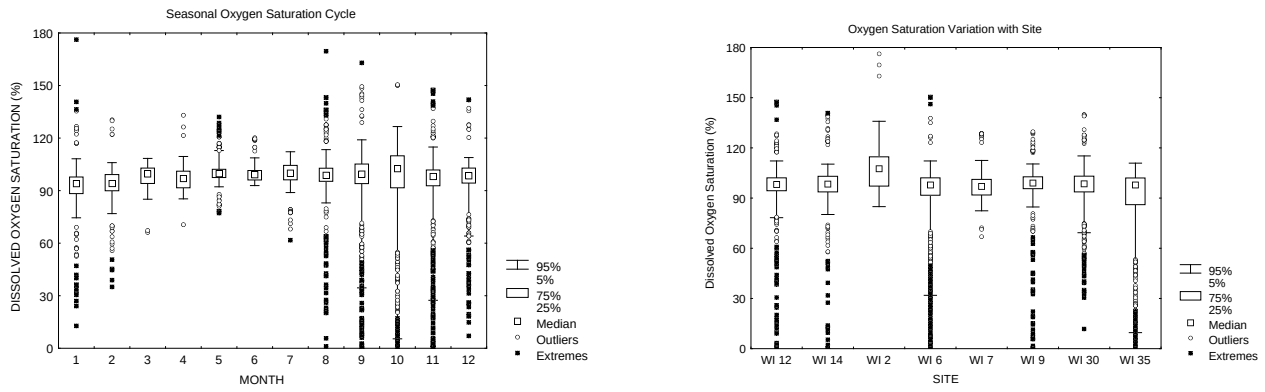


Figure 5. Seasonal dissolved saturation oxygen and variation between sites

Dissolved Oxygen Concentrations below 40% saturation (approximately 3 mg/L) were reported at all sites except WI 2 and WI 7 between August and February. When this occurred the deoxygenated zone rarely exceeded 0.5 m above the bottom. For the sites within the western and eastern basins the number of times a site reported deoxygenated waters decreased with distance from the Inlet entrance. Sites WI 6 and WI 35 had deoxygenated bottom waters on nearly every sampling run between September and October, while sites WI 9, WI 12 and WI 14 had fewer such reports. During the period November to February, the deoxygenation events occurred with a lower incidence and the areas over which the events occurred reduced in extent, shrinking towards the western basin. By January, only WI 6 reported dissolved oxygen concentrations below 40% saturated. Site WI 30 behaved differently from the sites in the basins, as although deoxygenated bottom waters were reported, they occurred much less often and the extent of deoxygenation was less severe.

Severe deoxygenation, with oxygen levels below 10% saturation (approximately 1 mg/L), was reported in the bottom waters at all the basin sites at some time during August to November. During this period 7% of all readings from WI 6 were severally deoxygenated, 3% from WI 9, 4% from WI 12 and 2% from WI 14.

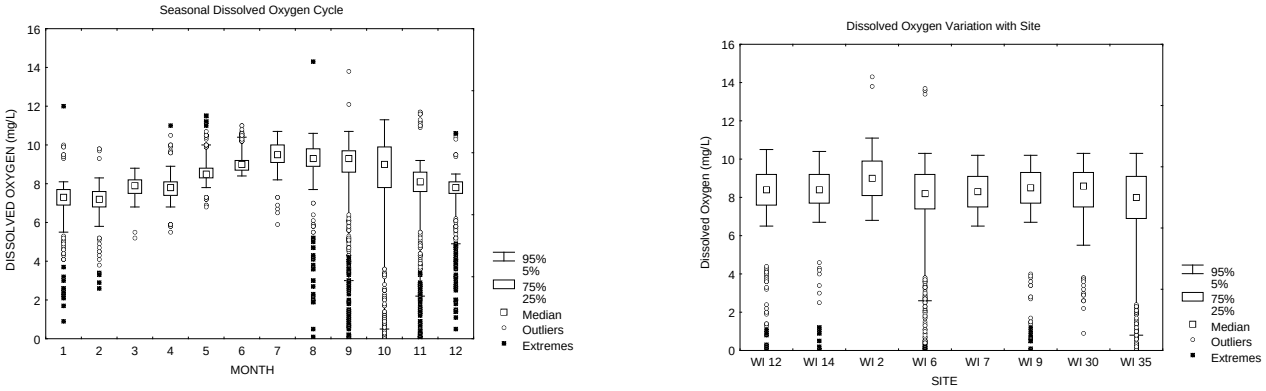


Figure 6. Seasonal dissolved oxygen concentration and variation between sites

Secchi disk readings

The secchi disk provides a measure of the transparency of water column and hence light penetration. The transparency declines due to algal blooms or suspended sediments. The secchi disk readings remained steady from January to July (Figure 7), with the secchi disk usually reaching the bottom at all sites (with a maximum possible reading of over 4 meters at WI 6). In August the median secchi readings dropped and reached a lowest level of about 1.5 m in September/October. In November and December the values started to rise back to the readings observed during the rest of summer and autumn. This behaviour was consistent between sites, with the secchi disk close to or on the bottom for the entire year at the shallower site WI 2. Sites WI 14 and WI 7 reported secchi readings of less than 1 m on several occasions, and WI 14 generally had the lowest secchi reading for each sampling run during winter.

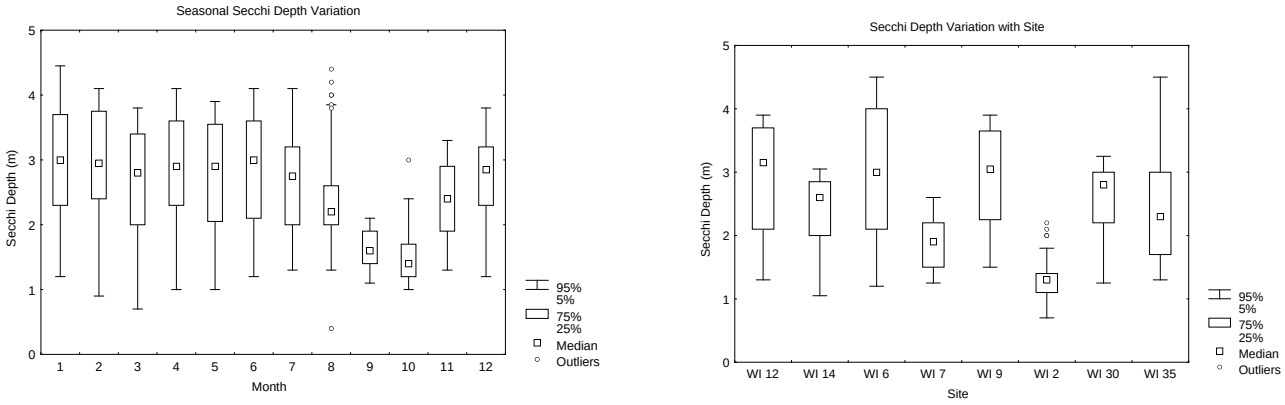


Figure 7. Seasonal Secchi disk depth and variation between sites

3.1.2 Water quality data

Ammonium

The variation in ammonium levels showed a seasonal dependence that was also tied to location and depth within the Inlet (Figure 8). For most of the year the ammonium levels in both the surface and bottom waters were low at all sites (but not always lower than the ANZECC, 1992 Guidelines for estuaries and embayments of < 0.005 mg/L). The annual median concentrations at each site were essentially identical, lying between 0.011 and 0.019 mg/L. The only elevated median was from WI 35 at a level of 0.029 mg/l (note that WI 35 was only sampled in 1997). All sites except WI 2, WI 30 and WI 7 showed a number of sampling points with concentrations exceeding 0.2 mg/L, considerably higher than the rest of the population. The occurrence of these extreme values was highly seasonal, being only reported between September and November (Figure 8), and as shown in Figure 10, occurred exclusively in the bottom waters when the sites were salinity stratified.

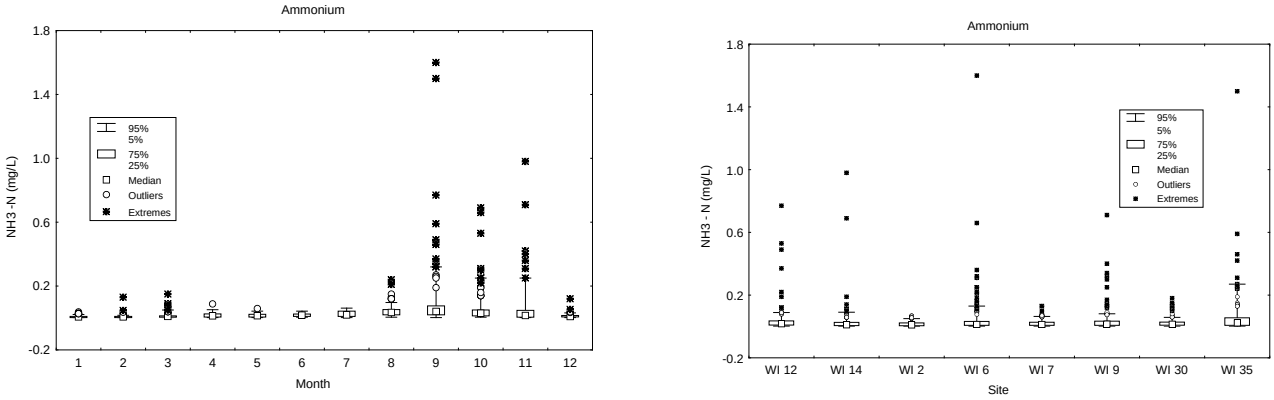


Figure 8. Seasonal ammonium concentrations and variation between sites

Nitrate and Nitrite (NOx)

Nitrate and nitrite are reported together as NOx, which is the concentration of nitrate plus nitrite. The behaviour of the NOx species was different from that of ammonium, as the concentration was principally dependent only on the season (Figure 9). For most of the year the NOx concentrations were very low at each site and depth. The annual median values were less than or equal to 0.011 mg/L for each site. For the three years from November to June all points fell below 0.05 mg/L, well within the ANZECC, 1992 Guideline for NO_3-N of 0.01 - 0.1mg/L. Over the period of July to August the NOx concentrations rose at all sites and depths, with the first observations of the rise and the highest values occurring at WI 14 and WI 12. The median value for NOx for August was 0.29 mg/L for WI 12 and WI 14 (with some values exceeding 0.4 mg/L), while the medians for the other sites were between 0.17 and 0.04 mg/L. The peak value occurred in August and over September and October the concentrations fell off rapidly, and by November the levels had returned to the values observed over summer.

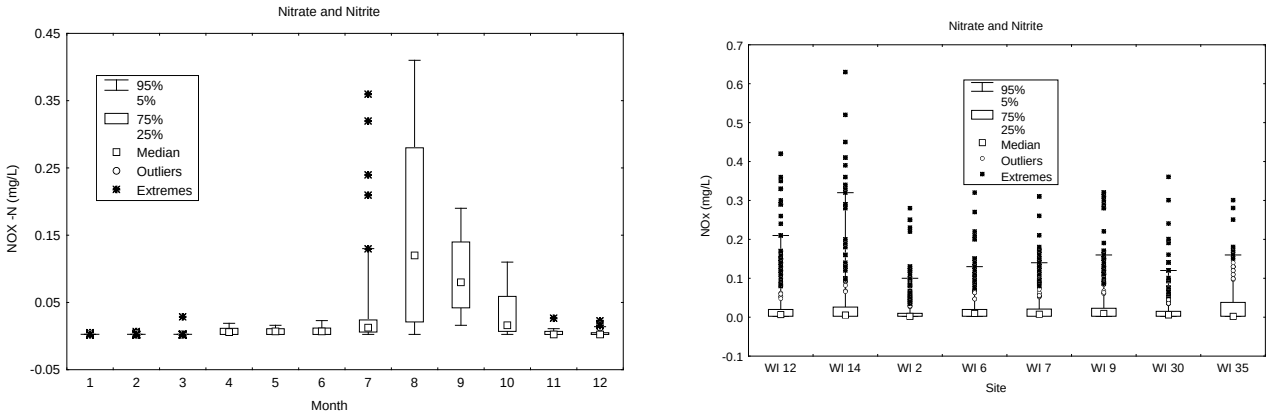


Figure 9. Seasonal NOx concentration and variation between sites

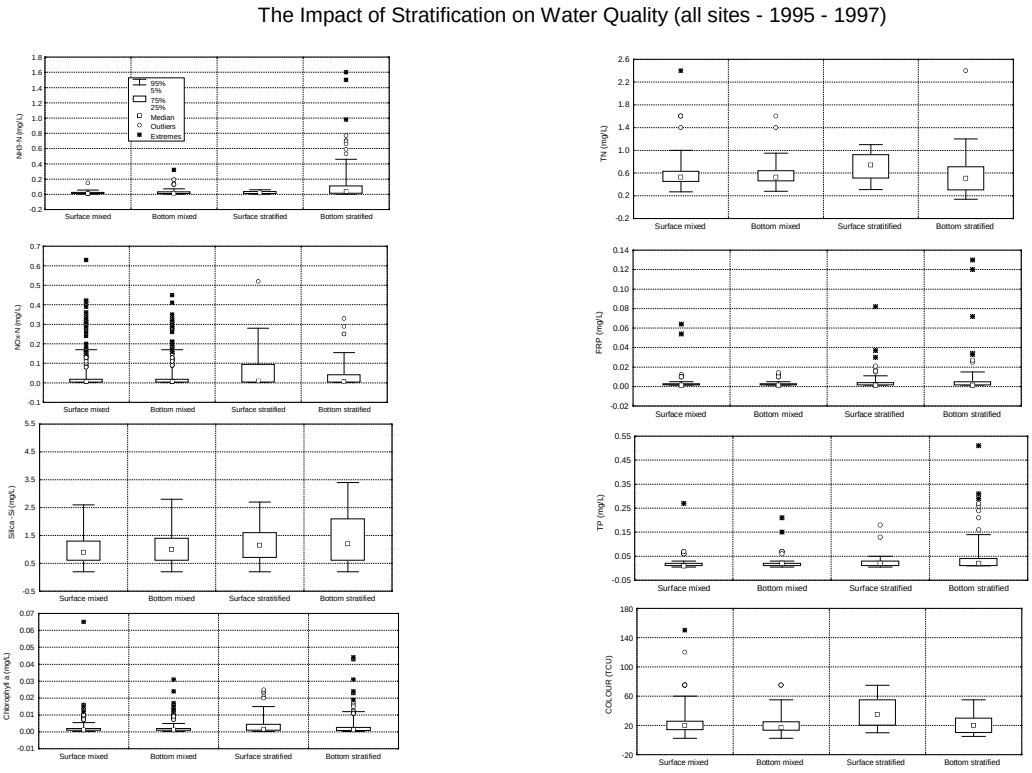


Figure 10. Data for all parameters categorised on location (surface or bottom) and the stratification condition at the time of sampling

A vertical salinity gradient of ΔS>5 ppt is categorised as a stratified condition.

Total nitrogen

Total Nitrogen (TN) is a sum of the individual nitrogen species, namely ammonium, NOx, dissolved organic nitrogen and particulate nitrogen, so the behaviour of TN is influenced by the behaviour of all the nitrogen species. The TN concentrations observed were similar across all sites (Figure 11).

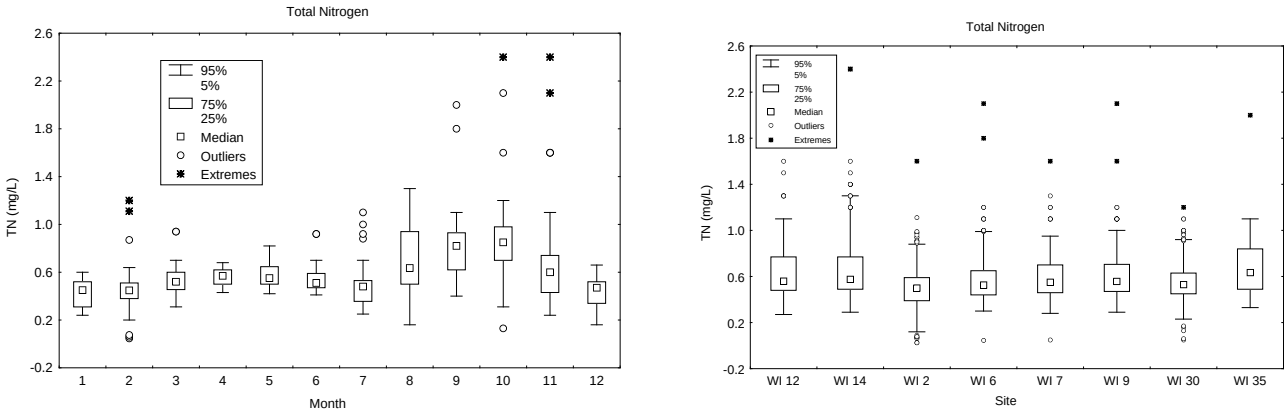


Figure 11. Seasonal TN concentration and variation between sites

The median values ranged from 0.50 to 0.68 mg/L, and each site showed a similar variance. Values greater than 1.2 mg/L, considerably higher than the rest of the population occurred between August and November, and were due in part to the high values of ammonium and NOx.

If the influence of these dissolved inorganic nitrogen species is removed from the measure of TN, the variation in the sum of dissolved organic nitrogen plus particulate nitrogen (includes contributions from phytoplankton) can be seen more clearly. Dissolved organic nitrogen plus particulate nitrogen was the most abundant form of nitrogen within the Inlet for most of the year (except when the concentrations of ammonium and/or NOx were elevated). The dissolved organic nitrogen plus particulate nitrogen concentrations observed were the same across all sites. A rise in the level of these species occurred in August, with a peak in October (with an October monthly median of 0.88 mg/L for the surface waters), which fell off by December to a median value of around 0.45 mg/L. The surface and bottom waters exhibited similar concentrations except when the dissolved organic nitrogen plus particulate nitrogen levels were elevated during the period September to October. During this time the surface waters tended to exhibit higher levels than the bottom waters (Figure 10).

Filterable reactive phosphorus (FRP)

Filterable reactive phosphorus concentrations within the Inlet were low for most of the year, except for some outliers or extreme values that were observed at some sites (Figure 12). The median FRP values for all sites were below the detection limit of 0.003 mg/L and 75% of all samples collected were below 0.004 mg/L (the ANZECC, 1992 Guidelines for PO_4-P for estuaries and embayments is 0.005 - 0.015 mg/L).

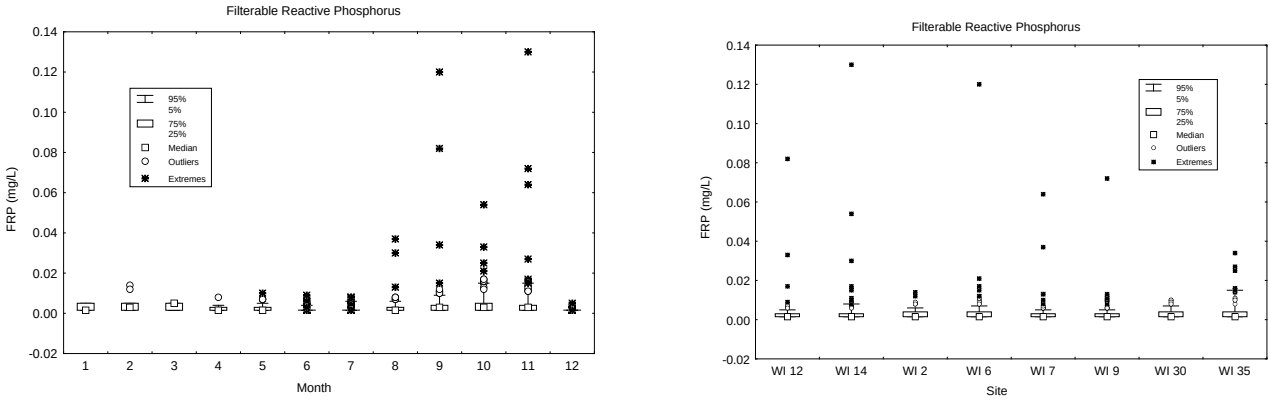


Figure 12. Seasonal FRP concentration and variation between sites

Concentrations above 0.005 mg/L only occurred between June and November, while concentrations above 0.02 mg/L occurred only from September to November and only at sites away from the ocean entrance (ie not at sites WI 2 and WI 30). Some extreme FRP values reported exceeded 0.05 mg/L. The

high FRP values occurred under two conditions; (1) in the surface waters (and not at elevated levels in the bottom waters) and (2) in bottom waters with a salinity stratified condition, when high ammonium levels (> 0.2 mg/L NH_4^+) were also reported (Figure 10). For both cases the high levels of FRP were only reported for a single sampling period, ie elevated FRP levels were not reported at the same site in the following sampling run.

Total phosphorus (TP)

Total phosphorus is the sum of FRP, dissolved organic phosphorus and particulate phosphorus, and as a consequence TP behaviour is influenced by each of its constituents. The TP levels were very low for most of the year, with sharp peaks of high concentrations occurring at certain sites under particular conditions (Figure 13). The concentration of TP in 75% of the samples collected was below 0.02 mg/L (the TP detection limit was 0.01 mg/L).

All sites, except WI 2 recorded a number of high values (above 0.1 mg/L) between July and November. If the influence of the FRP is removed from the TP values, the variation in dissolved organic phosphorus plus particulate phosphorus (which includes phosphorus from phytoplankton biomass) is apparent. The [TP-FRP] species also showed a seasonal variation with high values above 0.05 mg/L occurring several times per month from October to November, with the most extreme values recorded in November. By December the levels had fallen back to the baseline levels observed over the rest of summer.

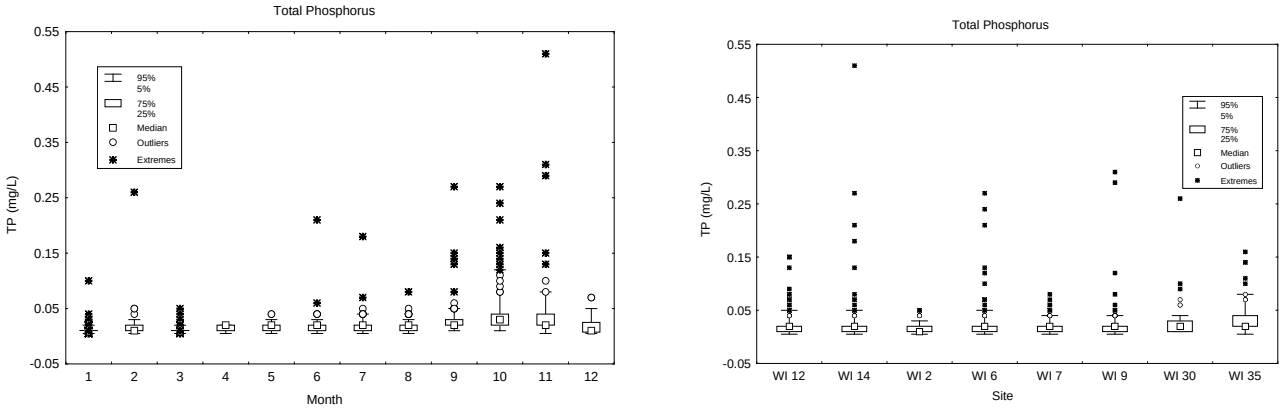


Figure 13. Seasonal TP concentration and variation between sites

The high values of [TP-FRP] occurred almost 4 times as often in the bottom waters, as in the surface waters. High levels in the surface waters were generally not coincident with high FRP values. Only once during the three year sampling did a high FRP level occur at the same time as a high [TP-FRP] within surface waters. This occurred at WI 14 on 25 October 1996 after an algal bloom was reported in early October. Elevated [TP-FRP] levels in bottom waters usually occurred under stratified conditions. High TP levels in bottom waters were generally due to high levels of [TP-FRP] and not high values of FRP. If high FRP levels in the bottom waters were reported, high levels of [TP-FRP] also occurred.

Silica

The silica levels showed stronger fluctuations over short time scales (weeks) than any of the other nutrient species measured. Unlike the infrequent high values of ammonium in bottom water and the more gradual variations observed in NOx, both of which occurred only between July and November, the silica levels fluctuations occurred over the whole year and the scale of the fluctuations ranged from a single week to several weeks. The fluctuations were observed at all sites, and the median silica levels were similar at all sites, with values ranging from 0.8 to 1.3 mg/L. Only site WI 2 showed levels lower with a median of 0.55 mg/L. An annual silica trend was evident, imposed over the short term fluctuations (Figure 14). An increase in the silica level occurred in July/ August, with a peak in September, followed by a rapid decline and return to low levels by December. Both the surface and bottom samples showed similar behaviour (Figure 10).

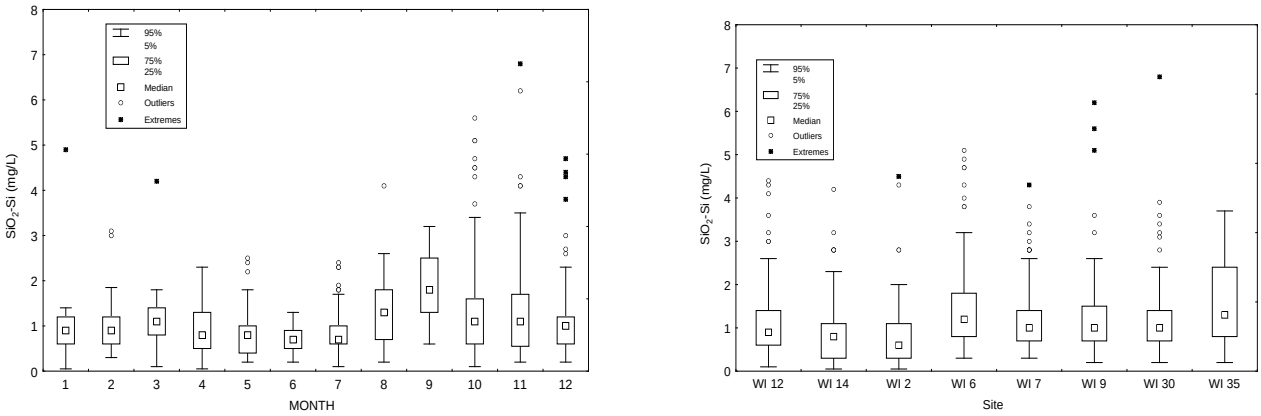


Figure 14. Seasonal silica concentration and variation between sites

Chlorophyll

The median chlorophyll levels for the three years of sampling data ranged from 1 to 1.7 ug/L, with WI 2 and WI 14 recording the lowest and WI 35 the highest median (Figure 15). The range of values experienced at each site were similar, with upper quartile values of between 1.7 and 2.1 ug/L for all sites, except WI 35, which had an upper quartile value of 3.8ug/L.

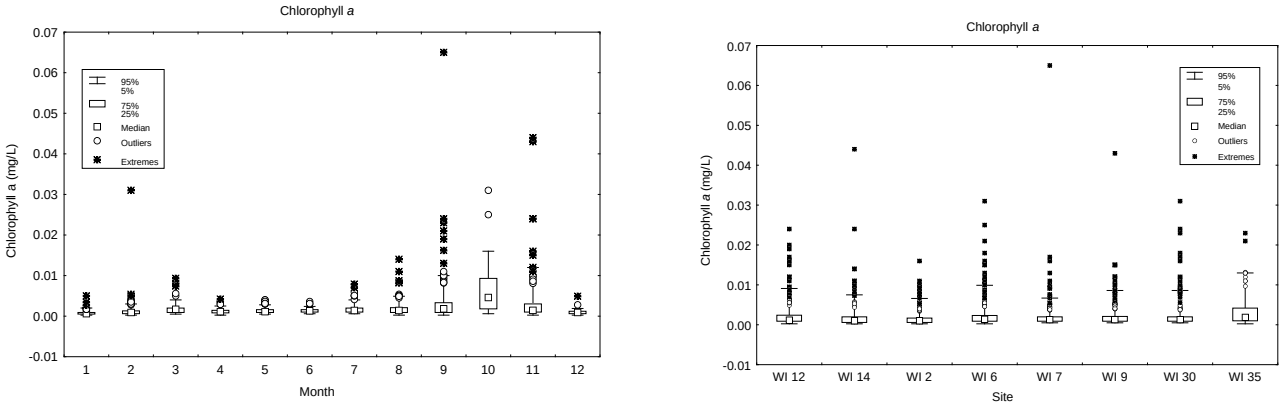


Figure 15. Seasonal Chlorophyll *a* concentration and variation between sites

The annual chlorophyll behaviour was characterised by periods from December to June when the values were very low, close to or below the detection limit of 0.5 ug/L. In July and August, the majority of samples remained low in chlorophyll, however a number of higher readings were reported. The monthly median levels rose over September, reaching the highest value in October.

Through August to November extreme values of 10 ug/L and above, considerably higher than the rest of the population were reported at all sites. The extreme values were often associated with a rise and a decline of chlorophyll levels over several weeks. The high levels recorded in August occurred in the surface waters and not in the bottom waters. During September and October high values were recorded in both surface and bottom waters, but the chlorophyll levels in the surface waters were usually higher than those in the bottom waters. The median chlorophyll for surface waters in October was 7 ug/L and 3 ug/L for bottom waters. In November the two depth levels had the same median, but the bottom waters reported several extreme values (up to 40 ug/L chlorophyll a), while none were reported in the surface waters. For the rest of the year the surface and bottom waters behave similarly.

Total organic carbon (TOC)

The median TOC levels reported at each site over the three year sampling period ranged from 7 to 8.5 mg/L, with WI 2 recording the lowest median and WI 35 the highest (Figure 16). All sites showed a similar range of values. TOC levels varied seasonally. From January to July the values were steady with monthly medians ranging from 7 to 7.5 mg/ L and a low variance in the data reported. An increase in the overall median and variance occurred in August, with a peak in October (October median was 10.8 mg/L), with a return to lower levels by January. The surface and bottom waters displayed no difference in behaviour.

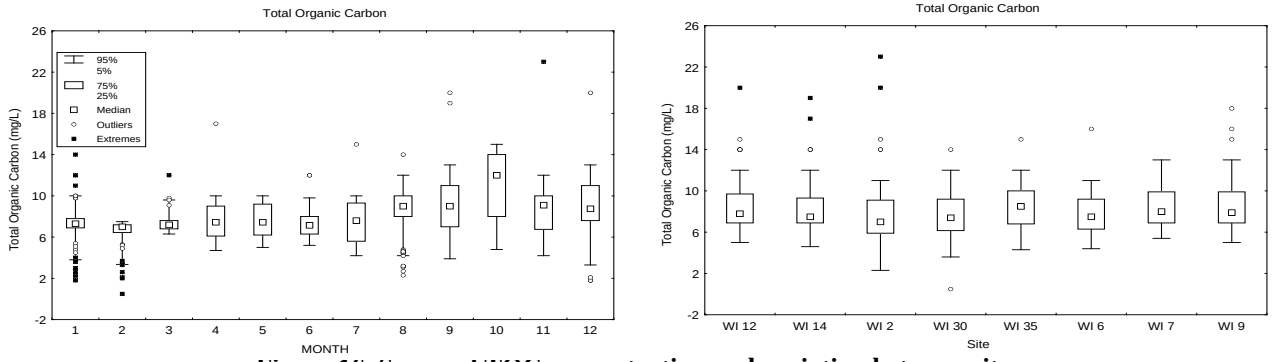


Figure 16. Seasonal TOC concentration and variation between sites

Colour

Two methods were used to determine colour over the three year sampling period. Until October 1996 colour was determined by a visual comparison method and reported in platinum cobalt units (PCU). After October 1996 the Gilvin colour comparator method using a wavelength of 440 nm was used to report colour. These data are shown in Figure 17 and Figure 18 respectively.

The values from each method are not interconvertible, however both data sets indicate the same behaviour. Sites WI 2, WI 30 and WI 6 have medians slightly lower than the other sites. The distribution of the data depends strongly on the location of the site. WI 2, WI 30, WI 6 and WI 35 have the smallest range in the data, whilst the other four sites have both a wider quartile range and total range. Extreme values were recorded from sites WI 14, WI 12, WI 7 and WI 9. The extreme values occurred between August and October, when higher median values were also reported. By December the colour levels had dropped and remained low, with little variance until July. The surface and bottom waters showed no difference in colour levels when the Inlet was mixed, however once the Inlet was stratified the surface waters had a slightly higher level of colour than the bottom waters.

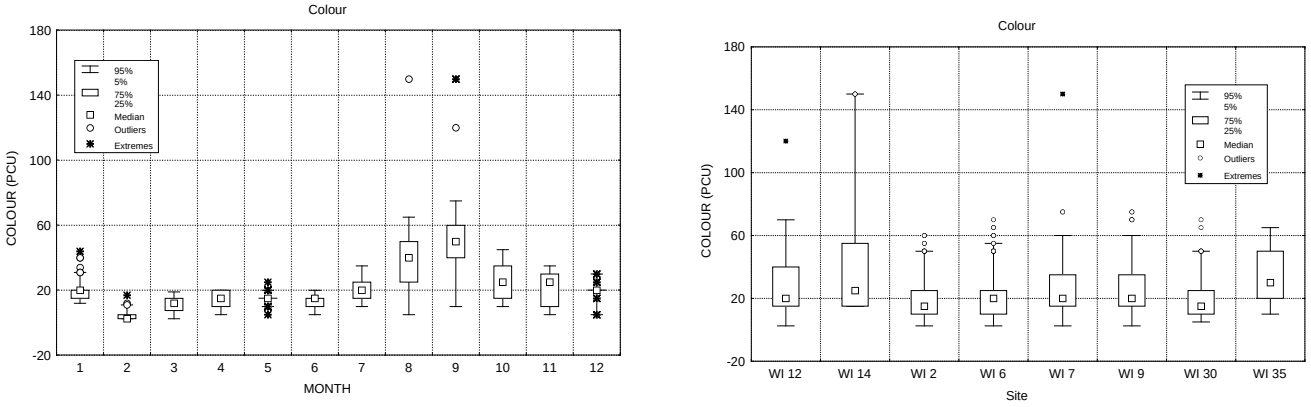


Figure 17. Seasonal Colour (PCU) level and variation between sites

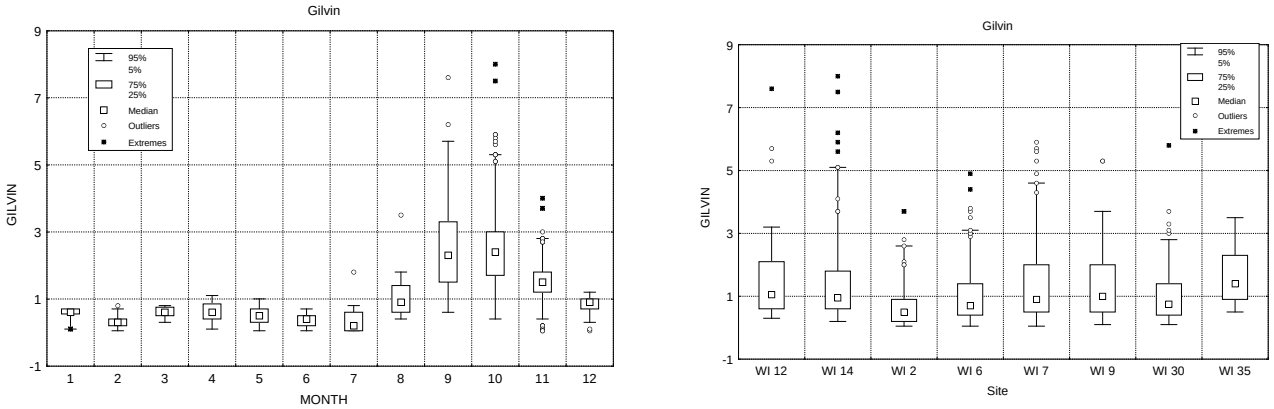


Figure 18. Seasonal Gilvin level and variation between sites

3.1.3 Phytoplankton

Phytoplankton cell counts

The total phytoplankton cell counts showed a strong seasonal dependence, with very little variation across sites (Figure 19). Between January and September the cell counts were low, with monthly median counts of around 1,000 cells/ml. Cryptophytes were the dominant species during this time, with low cells counts of diatoms, dinoflagellates and chlorophytes. A rapid rise in cell counts occurred in October, with a median of 19, 000 cell/ml and counts up to 450, 000 cells/ml. The cell counts in October and November reached bloom proportions and in all three years consisted primarily of diatoms and dinoflagellates.

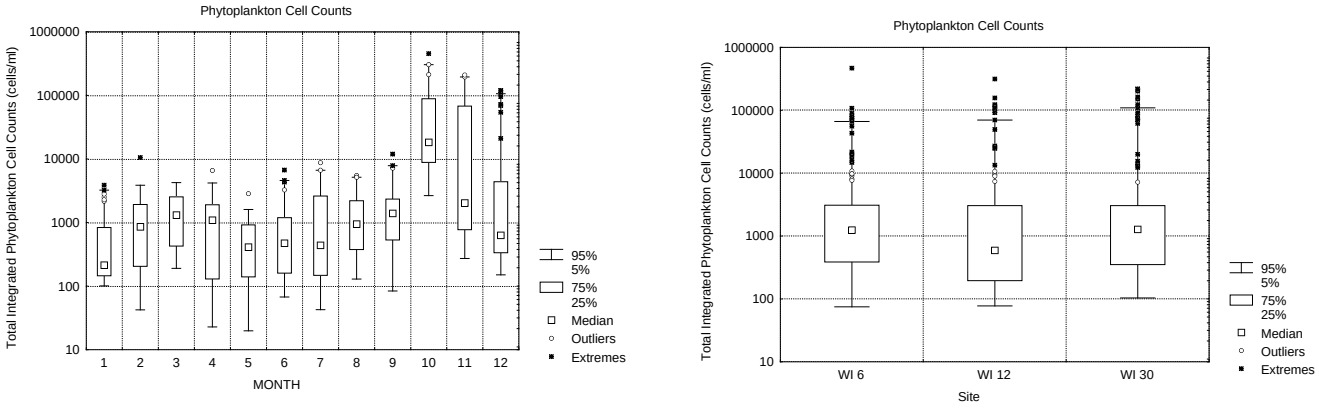


Figure 19. Seasonal phytoplankton cell counts and variation between cites (Note the logarithmic scale)

4. Discussion

4.1 Annual hydrodynamic cycle

From the previous section it is obvious that nearly all the important nutrient species in the Inlet showed a highly seasonal response. Prior to discussing the behaviours and relationships that lead to such patterns, it is necessary to describe the seasonally changing physical environment of the Inlet. This environment is highly variable and has the potential to strongly influence the fate of any nutrients. The following section describes the inflows and outflows of water to the Inlet and the resulting large scale water movements and associated physical conditions.

Previous researchers have described the hydrodynamics of the Inlet (Marshall, 1993 and Ranasinge and Pattariatchi, 1997). A summary of their findings was given in Section 1.1. It is a valuable exercise to assess the Water and Rivers Commission data in terms of the Inlet hydrodynamics, as the data now available is at a higher temporal resolution (weekly as opposed to quarterly) and includes dissolved oxygen as an additional parameter.

The discussion below is a detailed analysis of the physical data for the year 1996/97. Selected transects are presented to illustrate the important conditions of the Inlet. The transect used to contour the data is roughly an west-east alignment and is shown in Figure 20. The conditions reported in this year are described in terms of the states of the Inlet based on those proposed by Marshall (1994). Modifications to the hydrodynamic states described have been made due to the additional data that is now available. As each year has different characteristics the Section 4.1.5 is a brief discussion on the differences in the hydrodynamics observed between the three years.

Figure 20. Wilson Inlet showing the transect used in the contour plots of the physical data

4.1.1 The closed-dry state

After the entrance to the ocean had closed on 16 February 1996 and water exchange with the ocean was prevented, the salinity variation over the entire Inlet was less than 0.5 ppt with an average of 26.5 ppt. At each site the vertical variation in salinity was minimal (< 0.2 ppt). The oxygen levels were close to 100%, also with little variation across the Inlet. This well mixed condition persisted throughout the rest of summer and most of autumn. A representative set of transects for the closed state is shown in Figure 21.

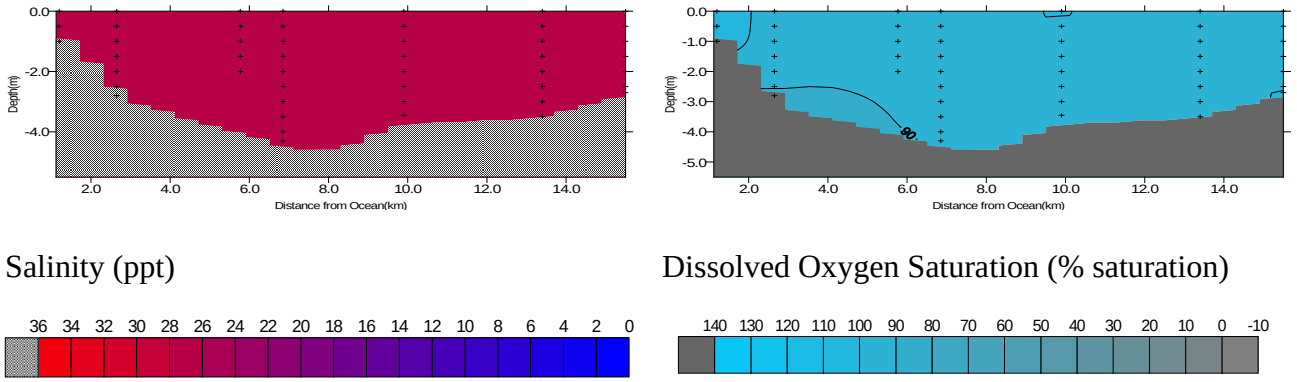


Figure 21. Salinity and dissolved oxygen transects for 21 March 1996, illustrating a typical closed summer well mixed state

The average salinity in the Inlet rose until July. The increase was 1.2 ppt in the first month following the closing of the bar, the rate of increase then slowed considerably, with the average salinity reaching a maximum of 28 ppt. The Inlet wide increase in salinity occurred first at the eastern stations (WI 9, 12 &14). These stations often displayed salinity values 0.5 to 1 ppt higher than those observed at the western and central sites. This may be due to the effect of evaporation being more pronounced in the shallower areas, or due to the groundwater flux from the saline aquifer in the eastern basin (Yu, 1998).

4.1.2 The closed-wet state

The first evidence of freshwater runoff reaching the Inlet was recorded on 24 May 1996. Site WI 7, the site closest to the Denmark River recorded a lens of fresher water about 1 metre deep, with a surface salinity of 23.8 ppt. The sampling run four days later found no evidence of this fresh water lens around WI 7. However, runoff from the eastern rivers was evident with site WI 12 having a surface salinity of 26.3 ppt. Again the fresher water lens did not persist, as it was not observed during the following sampling run 9 days later. This return to a homogeneous condition is probably due to wind mixing or diurnal overturning breaking down the thin layer of fresher water (Ranasinge and Pattariatchi, 1997). Due to the lack of rainfall in late May there was no river flow to sustain the fresher water lens against these mixing forces.

The importance of the different mixing processes in producing the observed salinity structures can be assessed using formulations developed by Spiegel *et al.*, (1986). The depth of the mixed layer due to wind mixing alone is given by:

$$h = \left(\frac{C_F C_N^3 U_*^3}{(C_F + C_E) N^2} \right)^{\frac{1}{3}} t^{\frac{1}{3}} \qquad (1)$$

where C_F=0.25; C_N=1.33; C_E=1.15, are the coefficients of energy conversion; U_*=shear velocity,

$N^2 = -\frac{g}{\rho o}\frac{d\bar{\rho}}{dz}$, the buoyancy frequency; t=time and h=

depth of mixed layer. The shear velocity U_* can be calculated from $U_*=(\tau/\rho_w)^{1/2}$ where τ is the surface stress and is given by $\tau=C_D\rho_A U^2$, ρ_w, ρ_o and ρ_A are the densities of water, a water reference density and air respectively, C_D is the Drag coefficient and can be approximated as 1.3×10^{-3} (Fischer *et al.*, 1979), and U is the wind speed at an elevation of 10m.

Using Equation (1) the winds which blew for over 15 hours, with an average speed of 8 m/s, (with gusts of up to 15 m/s) on May 30 1996, two days after sampling would have created a mixed layer 1.7 m deep in a waterbody with an original density difference of 1.6 kg/m^3 (which was reported at site WI 12 on May 28th). As the entire water column at WI 12 was mixed (a depth of 3.7 m) the strong gusts of wind (up to 15 m/s during the 15 hours) and convective cooling must have contributed to the mixing.

Sustained rainfall in 1996 commenced on 15th June and continued heavily for almost a month. The Inlet salinity levels remained stable for two weeks after the start of the winter rains. The sampling run on July 3rd gave the first indication of a fall in salinity values. The median salinity had fallen by nearly 2 ppt to 26.3 ppt but the Inlet was well mixed both vertically and laterally. This indicates that an efficient vertical and horizontal mixing regime was operating in the Inlet.

This contrasts with the effect of an the initial winter freshwater inflow in the Swan River (Stevens and Imberger, 1996), which results in fronts of fresher water flowing downstream. Slight stratification due to freshwater inflow was recorded on the following sampling run on July 10 (with a daily flow volume of 0.4 million cubic metres, and a total volume of 2 million cubic metres since the last sampling), with the 26 ppt isohaline pushed down to 2 m depth at the eastern sites (WI 12 and WI 14) and lying at 1 m depth over most of the western basin.

By the next sampling run 13 days later (July 23rd), with continued river flow (9 million cubic metres since the last sampling) the salinity over the entire Inlet had dropped below 26 ppt. All but the eastern-most site WI 14 was well mixed at around 24.3 ppt. At WI 14 the surface salinity had fallen to 20 ppt and the fresher water layer extended through the entire water column. As the fresher water layer was no longer evident at any of the other sites, considerable mixing must have occurred again. Equation (1) indicates that with the reported average wind speed of 5 m/s, wind mixing alone was unlikely to have mixed the water column, indicating the importance of turbulent mixing, longitudinal mixing and/or convective mixing. This agrees with Ranasinge and Pattariatchi, (1997) who calculated that overnight penetrative mixing was the major mechanism causing mixing of the Inlet during winter.

The Inlet salinity continued to fall, due to river inflow, with an average salinity of 22 ppt recorded on July 31 and 20 ppt recorded on August 6. The freshwater inflow continued to be efficiently vertically mixed with the Inlet water, as the salinity at all sites except the two easternmost sites was uniform with depth. On July 31 a longitudinal salinity gradient was set up across the eastern basin, with a difference of 3 ppt from site WI 14 to WI 9. By August 6 sufficient freshwater had entered the Inlet (30 million cubic metres in the past week) so that a freshwater plume could be maintained against the mixing forces. This plume extended to WI 12, with a surface salinity of 12 ppt at WI 14. The western basin was always more saline than the Eastern basin but continued to fall in salinity while remaining well mixed.

4.1.3 The open-wet state

The sand bar was breached on 7 August 1996 when the water level in the Inlet reached over 1 m AHD. For a month following this event a series of more frequent physical data sampling transects were conducted to study the impact of salt water entering the Inlet.

The first sampling run was conducted 4 days after the bar was breached, at the time of spring tides. By this time saltwater was detected at WI 2 and at WI 30, indicating that the saltwater had penetrated over the delta (Figure 22). Two sets of transects were collected during this sampling run, so giving more information on the progression of the saltwater. The second transect, conducted approximately 5 hours after the first transect indicated that the depth of the salt layer had increased and the salinity risen in this time. The first transect recorded only saltwater in the bottom reading at WI 30, with a salinity of 27 ppt. By the second transect, the bottom salinity had increased to 33 ppt and the saline layer was nearly 1 m thick. The penetration of the saline water observed in 1996 was comparable to that recorded in 1994 (Ranasinge and Pattariatchi, 1997) 1 week after the bar opened and in 1991 on the transect collected 13 days after the bar opened at high tide (Marshall, 1993).

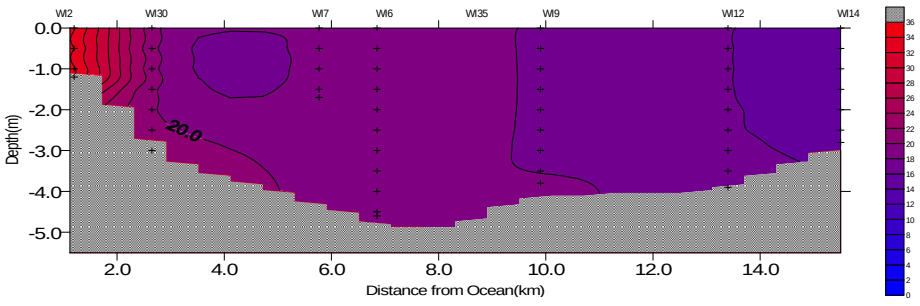


Figure 22. Salinity transect conducted on 12 August 1996 on a high tide

The high tidal levels associated with the spring tides continued for the 4 days following the 1996 bar opening. By August 15, 8 days after the bar opening, a saltwater layer 50 cm thick was detected at WI 6 and at WI 34 (see Figure A1 for locations of non-routine sampling sites). A sample collected at a location halfway between WI 34 and WI 7 showed no evidence of saltwater, indicating that the saltwater intrusion had not spilled out of the deeper region of the western basin. The daily stream flow on 15 August was 34 m^3/s. Marshall (1993) calculated that saltwater would penetrate the Inlet when the total streamflow into the Inlet fell below 35 m^3/s. The open winter state (no salt water retained in the Inlet over a tidal cycle) proposed by Marshall (1993) lasted for less than 8 days in 1996.

While the salinity at the western end of the Inlet began to rise, the continuing freshwater inflow resulted in the freshwater plume in the eastern basin deepening and extending seaward. By August 15 the freshwater filled the entire eastern basin and was recorded at WI 6 and WI 35 to a depth of 4 metres.

Over the next month the Inlet behaved as a typical salt wedge estuary (Pritchard, 1967) as described by Marshall (1993) with seawater progressing upstream under the influence of the tidal cycle and the freshwater outflow proceeding to the coast as a gravity current. As a consequence a very strong stratification was setup in both the eastern and the western basin. Ranasinge and Pattariatchi (1996) did not observe such a state, but did record a high salinity water present in the deeper part of western basin that did not move during the tidal cycle.

The wind strengths over this time were insufficient to mix a density difference of 10 kg/m^3. Convective mixing, indicated by Ranasinge and Pattariatchi (1996) to be the most important mixing mechanism in winter is described by Spiegel *et al.*, (1986) as

$$h = \left(\frac{4\alpha g H C_F}{(C_F + C_E) C_p \rho_o} \right)^{\frac{1}{2}} t^{\frac{1}{2}} \qquad \textbf{(2)}$$

where α is the expansion co-efficient for water = 2.57 ×10-4 /^{o}C, and Cp= specific heat for water = 4179 J/kg/C and H is the net heat loss.

If the net heat loss overnight is estimated at 200 Wm^{-2} and acted over 12 hours, the depth of the mixed layer would be only 0.4 m with an original density difference of 10 kg/m^3. Thus when the Inlet is strongly stratified convective cooling cannot destratify the water column.

The lack of efficient de-stratifying mechanisms is evident from the physical monitoring as the western basin was constantly stratified (based on the weekly sampling), with a salinity difference of 15 ppt between the surface and bottom waters. Marine water that had penetrated into the western basin was evidently restricted from flowing out on the ebbing tide due to the shallow delta. The depth of the saline layer can be approximately tracked by following the 20 ppt isohaline (any level higher than 20 ppt was due to inflow of saline water, as the Inlet was below 20 ppt when the bar was opened). This indicates that the saltwater intrusion generally filled the lower basin to a depth of 1 metre.

The progression of the saltwedge is difficult to accurately follow with the sampling frequency that is available. It can be seen, however, that the saltwater had moved from WI 30 on August 12 to WI 6 on August 15, a distance of 4.2 km. There were spring tides were over this time. The saltwater was not detected in the eastern basin until 27 August (the previous sampling having been on 22 August), when a bottom salinity of 21 ppt was recorded at WI 9 and WI 12. The next sampling run on September 2 reported a deeper layer, with the 20 ppt isohaline at a depth of 3.5 metres across the flat bottom of the eastern basin, and a bottom salinity of 27 ppt at WI 12. The slower progress of the saltwater in reaching the eastern basin is due to the occurrence of small tidal amplitudes associated with neap tides and the deviation that the saltwater front had to make around Pelican Point in order to reach WI 9.

4.1.3.1 Tidal dynamics

Heavy rainfall fell in mid-September, so that considerable river discharge continued to enter the Inlet. The importance of the freshwater overflow and tidal dynamics on the circulation of the system is highlighted by comparing the salinity profiles of 2 September (Figure 23) and 9 September (Figure 24). The 2 September transect was performed as the tide ebbed. This is reflected in the isohalines, which are stretched horizontally, towards the entrance to the ocean, and show 17 ppt water reached the Inlet mouth. A second vertical transect conducted at WI 2, 4 hours later indicated that no saltwater was present at WI 2 and that Inlet water filled the entire water column and

thus most likely flowed out of the Inlet. The 9 September transect was conducted at high tide, and the seawater can be seen moving into the Inlet as a vertical front, and plunging down the sill into the basin. Saltwater of salinity 35 ppt extended the depth of the water column at WI 2 and a saline layer slightly deeper than previously recorded filled the western basin. Compared to the profile on 2 September the 18 ppt isohaline was pushed back to WI 34 and lifted in response to the incoming tide. Opposing fluid motions occurred as the buoyant freshwater plume moved seaward against the incoming, tidally driven motion of the seawater, so that it appears that as the tide rose freshwater outflow to the ocean was prevented, so that flushing of fresher water from the Inlet was restricted.

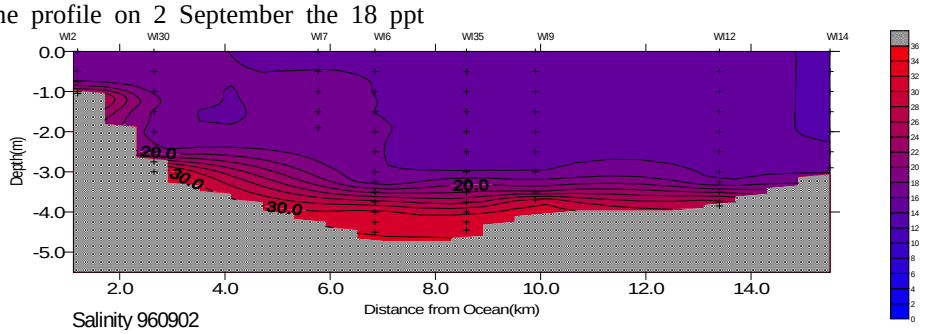


Figure 23. Salinity transect for 2 September 1996, conducted on the ebbing tide

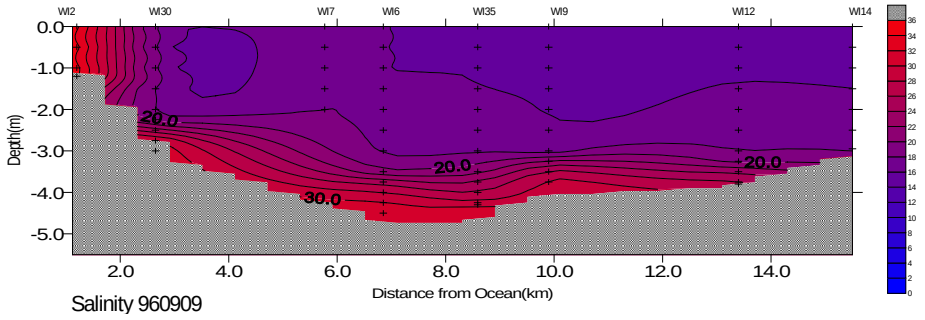


Figure 24. Salinity transect for 9 September 1996, conducted on the high tide

4.1.3.2 Winter mixing events

Despite the severe stratification that had set up during the spring tides, the sampling run conducted on 13 September indicated considerably less stratification than had been recorded on the previous run. The layer of saltwater that was present in the eastern basin was gone. A strong freshwater plume to a depth of 1.5 m was evident at WI 14, however the remainder of the eastern basin was close to well mixed, with a salinity of around 17.5 ppt. The low level of the low tide 6 days earlier combined with the freshwater discharge may have been responsible for barotropically advecting most of the saltwater layer from the eastern basin. Wind and convective mixing would have mixed the water column (but not completely) or wind induced currents may have longitudinally mixed the basin or moved the saline layer away from the sampling sites. The salt layer was obviously mixed with the above lying fresher water to some extent as the salinity of the water column at all sites was higher than previously reported. A salinity increase was found at sites WI 9 and WI 12 of greater than 1 ppt and the waters above the halocline in the western basin had increased by about 2 ppt.

The next sampling run, performed on 20 September showed further breakdown of the stratification and most of the Inlet in a vertically well mixed condition. In the western basin, some mixing of the saline layer must have occurred as there was an increase in salinity of the water column. Only the bottom 30 cm at sites WI 30 and WI 3 showed evidence of remaining saltwater. No saltwater was detected at the sites further from the Inlet entrance. Equation 1 indicates that the strong winds that blew on the 16 and 17th of September, with an average windspeed of 11 m/s for 12 hours, and with gust of up to 20 m/s were sufficient to have mixed the water column.

4.1.3.3 Re-establishment of stratification

The sampling in late September and October documents re-establishment of stratification within the

western basin, and indicates a similar series of events to those observed in September. Again the western basin stratified first, across the flat portion of the basin. The saline layer was recorded as only a thin bottom layer, often only 40 cm thick. Saline water was not detected in the eastern basin, at site WI 9 for 23 days, a similar delay occurred when the salt water first moved up the Inlet in August and September.

While the bottom layers of the western basin were consistently saline the waters above the halocline continued to fall in salinity over October. A freshwater plume was evident at site WI 14 on 2 October, with the 12 ppt isohaline extending just beyond WI 12 and reaching a depth of 2 metres. By 9 October, an additional 5.4 million cubic metres had flown into the Inlet and the 12 ppt isohaline had moved approximately 8 km downstream and had deepened to 3.5 metres. As with the previous months profiles the effects on the intertidal dynamics can be seen on the October profiles. The two transects conducted at ebbing tide during October (2 October and 30 October) show that the freshwater layer completely overflowed the delta and stopped seawater entering the Inlet. The 9 October transect, conducted at high tide shows the saltwater front moving in over the sill and filling site WI 30 to a depth of 1 metre with oxygenated seawater.

4.1.3.4 Oxygen dynamics

The oxygen levels showed a strong dependence on the salinity structure of the Inlet. The first significant salinity stratification, due to freshwater inflow ($\Delta S \sim 3$ ppt) was recorded on 6 August 1996 in the eastern basin, with the corresponding oxygen levels around 100% saturated. Five days later however, with the same level of salinity stratification, the oxygen levels in this region had fallen to less than 64% in the bottom 0.5 metres of the water column, the lowest levels that had been reported since the bar closed. The stratification in the eastern basin had been broken by the time of the next sampling and the oxygen levels had risen to around 100% again. The return to a deoxygenated state again was not detected until several days after salinity stratification was re-established.

The first observation of salinity stratification in the western basin on 15 August was also accompanied by low dissolved oxygen concentrations in the bottom waters. The stratification had at most persisted for only 3 days (time since the last sampling run) and in this time the oxygen levels in the lower 50 cm had reduced to 60% saturation. The following sampling runs revealed an a decrease to 0% saturation at WI 6 by 2 September 1996. The following sampling run on September 9 showed a slight recovery from the anoxic conditions, most likely due to the inflow of new marine water on a high tide, with the bottom waters reporting an oxygen saturation of 22%.

The oxygen level in the western basin remained low (below 35% saturation) below the halocline until the 20 September sampling, when the salinity stratification was destroyed due to the strong winds discussed previously. It can't be determined to what extent the deoxygenated waters were renewed with higher oxygen water in between sampling events or if the level remained low over the entire period. Analysis of the data from the salinity and temperature dataloggers deployed on the bottom of the Inlet will provide information on the long term extent of the deoxygenated layer. The oxygen levels of the bottom waters of WI 6 during this period are shown in Figure 25.

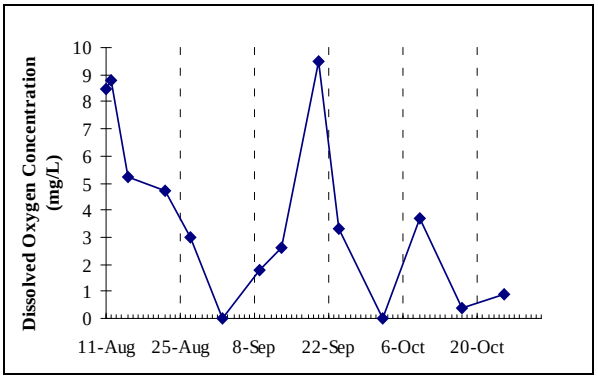


Figure 25. The oxygen level in the bottom waters of site WI 6

The western coastal sites, WI 2 and WI 30 rarely showed signs of deoxygenation, despite the severe (ΔS > 10 ppt) stratification that was often set up due to the inflowing marine water. This is because the waters were being renewed on a nearly daily basis with fresh, warmer, well oxygenated marine water.

The eastern basin had until August 27th been subject to mild salinity stratification, due to the buoyant freshwater plume. On 27 August, the first highly stratified conditions (ΔS >5 ppt) due to marine water intrusion was reported in the eastern basin. Co-incident with this was the dropping in dissolved oxygen concentration to below 2 mg/L at site WI 12 in the bottom 20 cm. By the next sampling 6 days later, sites WI 9 and WI 12 were anoxic. The subsequent oxygen

levels observed over the next few months were dependent on the destruction and re-establishment of the saline bottom layer in the eastern basin and on the strength of the salinity stratification due to the freshwater plume. When the water column was well mixed the oxygen levels throughout the water column were around 100%. However once a salinity stratification was set up the dissolved oxygen concentrations dropped; the extent determined by the strength and duration of the stratification. An example of the deoxygenation structure associated with salinity stratification is provided in Figure 26.

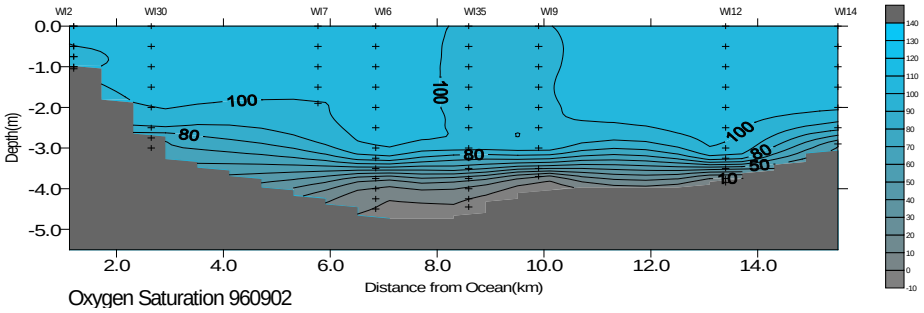


Figure 26. Oxygen saturation transect for 2 September 1996

While the waters below the halocline tended to be deoxygenated, the waters above were well mixed in regard to salinity and fully oxygenated. Often supersaturated surface waters were reported when the bottom waters were deoxygenated. The supersaturation is an indication of biological activity photosynthesizing and inputting oxygen into the water. The first widespread report of waters with dissolved oxygen concentrations greater than 110% saturation was on 20 September 1996, when the 110% saturation isopleth extended to a depth of 1.5 m across the western basin to WI 6 and northward to WI 7. High total integrated cell counts of > 2,000 cells/ml of diatoms and dinoflagellates were reported at WI 6 and WI 30 at this time. The following sampling run on 24 September showed the 110% isopleth was 2 m deep across the western basin, and 1 m deep to WI 12 in the eastern basin. The supersaturated water had extended further by the following run, and was reported at all sites, the associated integrated cell counts were greater than 100, 000 cells/ml at WI 30 and WI 12.

Pockets of supersaturated water were reported for several weeks after the Inlet wide event. A second large scale supersaturation occurred on 11 November, when again dissolved oxygen concentrations exceeded 110% saturation down to a depth of 3 metres, across the whole Inlet. The highest level reported was 147% at WI 12. The corresponding cell counts at this time at WI 12 was low compared to the high values recorded previously. This indicates that the magnitude of the supersaturation does not correlate well with the total cell counts. This may be because the elevated dissolved oxygen concentrations observed are dependent on the time of day and the amount of sunlight, and so do not necessarily represent the algae biomass adequately. The algae bloom was also probably heterogeneous both vertically and horizontally. The integrated sampling technique used should have smoothed out the vertical cell distribution, but a horizontal variation in cell densities could not be taken into account, so that the algae cell count collected at a single site may not be completely representative of the bloom.

4.1.3.5 Sediment oxygen demand

From the data in Figure 25 an apparent sediment oxygen demand (SOD) can be roughly calculated. Walker and Snodgrass (1986) considered the rate of oxygen consumption in aerobic sediments to be controlled by the oxidation of organic matter and modelled the process using Michaelis-Menton kinetics with an effective half-saturation constant of K_o=1.4 mg/L (which was found to be constant for all sediments studied). The rate of oxygen diffusion into the sediments was considered to be mass transport limited and modelled by Fick's Law of Diffusion across the sediment water interface. If as an approximation it is assumed that initially the aerobic SOD is the dominant term then

$$\frac{V}{A_s}\frac{d[O]}{dt} = -SOD_{\max}\frac{[O]}{K_o + [O]} \qquad (3)$$

Where SOD_{max} is the maximum aerobic oxidation rate per unit area, V is the volume of water and A_s is the area of sediment in contact with the V. If the volume of water V is taken to be $V=A_s h$, where *h* is the depth of

the lower saline layer, integrating this expression over two time periods t_1 and t_2, with corresponding oxygen levels of $[O]_1$ and $[O]_2$ gives

$$SOD_{\max} = -h.\frac{K_o(\ln([O]_2 - \ln([O]_1) + ([O]_2 - [O]_1)}{t_2 - t_1} \quad (4)$$

Using the oxygen levels associated with the first deoxygenation event at each site gives the values shown in Table 1. This value is only the apparent maximum SOD, as the oxygen consumption associated with the water column is also included in the calculation. The oxygen consumption due to nitrificatrion may have been an important component of the observed oxygen demand. Of course the calculation of SOD using weekly data is prone to error, for example underestimation would occur had the water column been refreshed inbetween sampling events. The selection of the depth *h* also introduces errors as the oxygen level in the bottom layer is not uniform, but varies with the lowest oxygen level at the bottom. Using *h* as half the depth of the saline layer, is an attempt to take this into account.

Table 1. Observed fall in oxygen levels with the first onset of stratification in 1996 and the calculated apparent sediment oxygen demand from equation 1

Site	WI 6	WI 35	WI 9	WI 12	WI 14
$[O]_1$ (mg/L)	8.8	7.9	7.8	8.3	8.9
$[O]_2$(mg/L)	3	1.9	3.8	1.9	0.9
Time (days)	15	11	5	12	8
2h (depth of saline layer)	1	1	0.75	0.6	0.5
SOD_{max} (mg/m^2/day)	245	365	370	210	350

From Table 1 it is evident that there was significant variability between some sites, but no distinct trend between the east and west basin. Douglas *et al*, (1996) conducted *in-situ* benthic chamber experiments at sites close to WI 6 and WI 9. The chambers used had a volume of 650 L and a sediment surface area of 0.78 m^2. The oxygen consumption rates were measured at 980 mg/m^2/day and 1220 mg/m^2/day for the two sites respectively. These values are up to a factor of 4 times greater than those calculated in Table 1. The discrepancies may represent differences in experimental and calculation method, seasonal changes and the inhomogeneous nature of the sediments. The values fall within a range recorded in other studies. Hamilton Harbour (a hyper-eutrophic embayement in Canada) had values at 16ºC between 1310-1970 mg/m^2/day while Chub Lake (an oligotrophic lake) recorded 271 - 284 mg/m^2/day (Walker and Snodgrass, 1986). The Wilson Inlet values seem considerably higher than those reported for the Swan River which lie between 15 and 266 mg/m^2/day (Ghisalberti, 1997), determined from laboratory core experiments.

4.1.4 The open-dry state

The effect of the decline in freshwater runoff and the continual inflow of seawater can be observed in the transects collected from November until when the bar closed in February. Over this period the salinity of the water above the halocline rose slowly as evaporation and mixing of the saline water with the remaining water column occurred. With the reduced rainfall and hence river flow, shoaling of the channel would have occurred, so reducing the tidal exchange. The transects collected over this time indicate that salt water continued to intrude into the Inlet despite the reduced channel dimensions. This observation is different from that of Ranasinghe and Pattriarchi (1994) who suggested that during the open summer condition there is no salt water intrusion into the Inlet. This conclusion was however made on only a single transect conducted one week prior to the closing of the bar. The inflow/outflow simulation developed by Boughton (1997) also indicates that marine water intrusion into the Inlet continues (but in reduced amounts) until the bar closes,

The western basin remained stratified until the sampling run on the 6 February 1997 (Figure 27). The stratification in the eastern basin appears to be more susceptible to longer term destruction, with a well mixed condition persisting over December. A slight stratification, with bottom salinities up to 28 ppt was re-established in the eastern basin in January. This indicates that saline water was still intruding up to 12 km from the ocean entrance, possibly aided by wind-driven bottom currents moving from west to east.

The eastern basin became well mixed on 30 January 1997, with a salinity of 25 ppt. The stratification in the western basin persisted until 6 February (when it was last observed). But due to the higher salinities of the water above the halocline and strong easterly winds continually mixing the water column, the stratification was not as ‘strong’ as that observed during September

and October, with a broad halocline and a vertical salinity difference of only ΔS = 3 ppt at WI 6 and ΔS = 5 ppt at WI 30. The last two transects collected before the bar closed (on February 23 1997) reported marine water at sites WI 2 (bottom salinity of 35 ppt) and WI 30 (bottom salinity of 33 ppt) but not at any sites further east. Due to the considerable shoaling of the Inlet entrance the seawater intrusion was of insufficient volume so that it was either flushed out during the ebbing tide or was completely mixed into the water column by wind mixing forces.

Again the presence of a seawater intrusion is in contrast to Ranasinghe and Pattriarchi (1994) who did not observe any progression of seawater at corresponding stage in 1994. This may be due to differing tidal amplitudes at this time between years. Three days after the bar closed there was less than 2 ppt difference across the entire Inlet. The higher salinities were recorded in the very bottom waters of WI 2 and WI 30. The average salinity was 27 ppt which was within 0.5 ppt of the average salinity recorded after the 1996 bar closing.

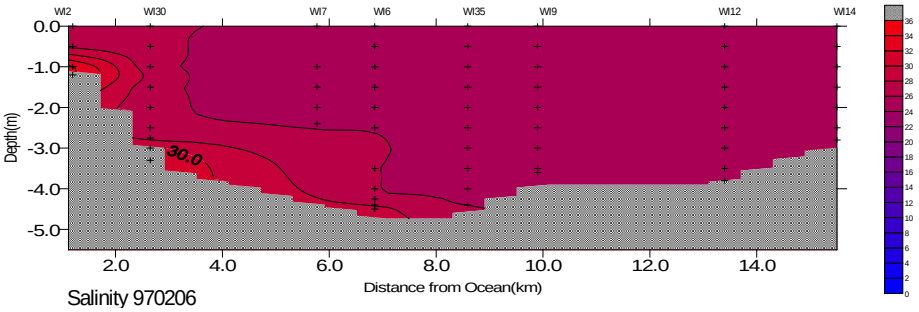


Figure 27. Salinity transect conducted on 6 February 1997

The oxygen levels continued to emulate the salinity structure, with deoxygenated water occurring below the halocline in the western and eastern basins. Oxygen supersaturated water was not reported after 19 November. As the severity of stratification lessened, the extent of deoxygenation also lessened. The coastal sites WI 2 and WI 30 were still being refreshed by incoming marine water, so low dissolved oxygen concentrations occurred much less frequently. Site WI 30 did experience some deoxygenation, with levels < 34% saturation in the bottom waters occurring at the same time as the western basin and site WI 9 and WI 19 were < 11% saturation. This severe deoxygenation occurred at the time of neap tides, when limited marine water was able to penetrate the Inlet and refresh the oxygen levels. The fact that site WI 30 was deoxygenated, indicated that the saline intrusion had not even penetrated to 2.5 km at this time.

The transect conducted on 26 February 1997, three days after the bar closure, showed deoxygenation at WI 30 and WI 2 in the lower waters below the very weak halocline, the bottom dissolved oxygen saturation being 35% and 70% respectively. This was the lowest dissolved oxygen concentration which had been reported at WI 2 over the past year, indicating how important the continual intrusion of oxygenated marine water was to maintaining the oxygen levels when salinity stratification existed. By the next sampling run the entire Inlet was well mixed with respect to salinity and fully oxygenated.

4.1.5 Interannual salinity variation

The annual salinity cycle observed in Wilson Inlet is very distinct and strongly influenced by the state of the ocean entrance, the tides, the freshwater discharge and wind events. As a consequence of these many highly variable factors it would be expected that the behaviour of the salinity structure each year would be different. The following section deals with the observed differences between the three years in an attempt to relate them back to the controlling variables and the hydrodynamics.

The time at which the bar closed varied over the three years of this study; the bar was reported to have closed in December 1994, February 1996 and February 1997 and February 1998. The salinities at key events in each year are given in Table 2, and it is evident that considerable variation occurred each year, even with the bar opened in the same location. A simple water and salt balance can be undertaken to assess the

different volumes of salt water that exchanged with the Inlet water.

Following the budgeting guidelines recommended for the Land and Ocean Interaction in the Coastal Zone Study (LOICZ) (Gordon *et al*, 1996), the following equation for the conservation of mass can be applied to any system:

$$\frac{dM}{dt} = \sum inputs - \sum outputs + \sum [\, sources - sinks \,] \quad \textbf{(5)}$$

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

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

$$\frac{dM}{dt} = \frac{dV}{dt} = 0 \quad (6)$$

where V is the volume of the system. There are no internal inputs or outputs of water, so that equation (6) simplifies to

$$0 = \sum inputs - \sum outputs \quad (7)$$

Considering all the inputs and outputs to Wilson Inlet, the water balance in equation (7) can be expressed as

$$0 = \sum (V_Q + V_P + V_G + V_R + V_x) + \sum (V_E + V_X) \quad (8)$$

where the following subscripts represent inputs and outputs from the following
Q - river inflow; P - precipitation; E - evaporation; G - groundwater
R - residual flow (the flow required in the system to keep the total volume constant)
X - exchange flow (inflow of seawater)

The assumption required to apply equation (6) to a system is that the water level is steady over time. Over a yearly cycle Wilson Inlet undergoes marked water level changes due to the opening and closing of the bar. Thus the assumption of a steady water level is not valid over short time frames in Wilson Inlet. It is thus necessary to apply equation (6) over an entire year, over which time the water level does remains essentially constant.

Considering equation (8) over a year enables the residual flow to be described as

$$V_R = -V_Q - V_p - V_G + V_E \quad (9).$$

The salt balance can be performed in a similar fashion, however to determine the exchange of saltwater over a year, the steady state assumption cannot be applied, rather equation (5) is used to give:

$$S_0V_{inlet} + S_QV_Q + S_RV_R + S_{X\text{-}in}V_X - S_{X\text{-}out}V_X + S_EV_E + S_GV_G + S_PV_P = S_1V_{inlet} \quad (10)$$

where S_0 and S_1 are the salinities each year just prior to the bar opening. Taking $S_Q \sim S_P \sim S_E \sim S_G \sim 0$, S_R and $S_{X\text{-}out}$ as the salinity of the outflowing surface layer and $S_{X\text{-}in}$ as the salinity of marine water, the exchange volume of seawater can be calculated for each year.

The values are shown in Table 2.

It is evident that the seawater exchange volume varied considerably over the three years. This is despite the fact that bar opening was conducted on the same side in each of these years. Other external (seasonal) factors have more influence on seawater exchange then the location of the channel to the ocean.

For example considerably less saltwater entered the Inlet in 1997, then in the preceding two years. Analysis of the waterlevel data at Albany shows that the ocean water level was lower in 1997 for much longer periods than in the previous two years. This was due to a combination of low tidal levels and persistent high pressure systems. As a consequence the marine water did not develop the elevation required to intrude into the Inlet on a regular basis.

As well as impacting on the overall salinity in the Inlet, the reduced saltwater intrusions led to the setup of weaker stratification than in the previous two years. In particular the eastern basin was less severely stratified and the stratification did not persist for as long. This is evident in Figure 34.

Table 2. Salinity levels (ppt) at key times for the 1995 through to the 1997 bar openings and the calculated volume of seawater exchanged

Opening Event	1995	1996	1997	1998
Post bar closure for previous years' opening	**21.2**	26.2	27	25
Maximum Salinity - bar closed period	24.7	28.1	27.8	25
Salinity (average) pre bar opening	15.4	19.9	22	**17.3**
Post bar closure	26.2	27	25	
Length of bar open period (approximate) (days)	180	200	154	
Total Riverflow between each bar opening (Mm^3)	100	183	153	
Calculated Seawater Exchange V_x (Mm^3)	124	175	101	

4.2 Nutrients

The chemical species that were analysed all showed a strong seasonal dependence, with low levels for much of the year but with a period of several months in which high concentrations were reported. The occurrence of these high concentrations often coincided with the occurrence of freshwater or stratification within the Inlet. There appears to be a strong relationship between the hydrodynamic state of the Inlet and the water chemistry.

With a highly seasonal physical and chemical environment it is not surprising that biomass in the Inlet also shows strong seasonal dependence. The timing of the annual phytoplankton bloom (that has occurred each year from 1995 to 1997) is related to the high levels of nutrients in the water column. The variation in abundance of the dominant macrophyte in Wilson Inlet, *Ruppia megacarpa* can partially be attributed to the physical characteristics (salinity, temperature, turbidity and pH) of the water column (Carruthers, 1997).

The following section explores in more detail the interaction between the physical, chemical and biological processes in the Inlet. This is done by explaining the chemical behaviour and determining the sources and sinks of nutrients within the Inlet by considering the hydrodynamics and biological activity.

4.2.1 Catchment nutrient input

Prior to discussing the water quality in the Inlet it is necessary to understand the external sources of nutrients and other species that are coming into the Inlet. The dominant source of external nutrients would be derived from the catchment and hence have a highly seasonal variation in input. The results from the catchment monitoring programme are discussed in detail in Donohue *et al.,* (1998). A brief summary of the results particularly important to the Inlet is provided below.

From the Wilson Inlet catchment monitoring it is evident that nutrients were exported from the catchment. The samples collected from the Sleeman River and Cuppup Creek reported TN levels above 1 mg/L, 95% of the time, and medians above 2 mg/L. The levels in the Hay River with a median around 1 mg/L, but high values above 3 mg/L were reported in June, July and August. The nitrogen species that make up the TN level have only been accurately measured since 1997. From the limited amount of data that has been collected it is evident that dissolved inorganic nitrogen (DIN, ie NOx + NH4) generally makes up only a low fraction of the TN in all the inflows (ie there were high organic nitrogen [TN-DIN] levels coming into the Inlet). NOx is a more significant species in the

Cuppup and Sleeman, contributing between 10 to 60% in the Cuppup and 10% to 20% in the Sleeman of the N through June to August, with levels declining over this period. The ammonium contribution was generally less than 10 % of the total nitrogen load, with high ammonium levels coinciding with the high NOx levels. The Hay River behaved slightly differently, higher levels of DIN only occurring between July and September, co-incident with higher TN levels

TP levels in Sleeman River and Cuppup Creek were also high, with medians above 0.1 mg/L. The other inflows reported low levels below 0.1 mg/L for most samples. The proportion of FRP was variable. Cuppup and Denmark had low proportions of FRP while Hay and Sleeman had higher levels of FRP, ranging from 15 to 100%. The water quality in the tributary inflows is discussed in more detail in Donohue *et al.*, (1998).

The highest flows occurred between July and October and the highest nutrient concentration in the waters were reported over June, July and August. It can roughly be assumed that the peak of the nutrient inputs to the Inlet occur in July and August, with lower level input sustained over September and October.

4.2.2 Salt, water and nutrient budget

To understand the nutrient dynamics of the Inlet, it is insufficient to know only about the concentrations and quantities of the nutrients coming in from the catchment. It is also important to have estimates of the nutrients leaving the system (in particular the nutrients exported to the ocean) and estimates of the nutrients that are either retained or released within the Inlet.

A water and salt balance is a seemingly straightforward method to determine such quantities. When the values calculated from the water and salt balances are coupled with measured nutrient concentrations, estimates of the loads of nutrients moving through the Inlet are obtained. The Land and Ocean Interaction in the Coastal Zone (LOICZ) steady state methodology (Gordon *et al*, 1996) has been applied to Wilson Inlet to obtain estimates of the outflow volumes and the sources and sinks with in the Inlet.

To obtain an average picture of how the Inlet behaved over 1995 to 1997 the LOICZ methodology is applied to the average data for these years. A steady-state assumption is applied to the Inlet over these three years so that equation (6) can be used. The steady-state assumption is not strictly valid, as evident in Table 2, however over 1995 – 1997 the inflow volumes were of the same order of magnitude, the factor most likely to cause deviations from steady state. The values calculated in this LOICZ model only provide indications of the relative magnitudes of each term.

Following on from equations (5) to (9) in Section 4.1.5 the steady state salt balance uses salt as a conservative material, so equation (5) simplifies to equation (7) if it is assumed that the mass of salt in the system remains constant over the time frame of interest. In order to perform the salt balance an average salinity must be assigned to each of the water inputs and outputs. This gives the following salt budget (the terms used are the same as in Section 4.1.5):

$$0 = \sum (S_Q V_Q + S_P V_P + S_G V_G + S_R V_R + S_{X-in} V_X) - \sum (S_E V_E + S_{X-out} V_X) \quad (11)$$

The quantities derived from the salt budget can then be used in a nutrient budget. The conservation of mass equation, equation (5) can be applied to any material. For materials which undergo net transformations within the system the Σ[sources-sinks] term in equation (5) is non-zero and represents the "non-conservative behaviour" of the system. Nutrients such as nitrogen and phosphorus are expected to display non-conservative behaviour. For a material Y equation (5) can be written as:

$$\frac{dVY}{dt} = \sum (Y_Q V_Q + Y_P V_P + Y_G V_G + Y_R V_R + Y_{X-in} V_X) - \sum (Y_E V_E + Y_{X-out} V_X) + \Delta Y \quad (12)$$

where $\Delta Y = \Sigma$[non-conservative sources-sinks].

Application of LOICZ budget to Wilson Inlet

For equation (6) to be valid it must be applied over an entire year, so that the water level is constant. The volumes of water calculated for each of the quantities in equation (8) are thus annual totals. The volumes and how they were determined are presented in Table 3 below.

Table 3. Summary of water volumes used for the water balance

Quantity	Value	Data Source
V_Q	156 million m^3/year	Average of annual total flows for 1995 - 1997. Sum of volumes measured from Hay, Sleeman, Cuppup and Denmark Rivers. Including an inflow factor of 3.5 × Sleeman volumes to account for ungauged catchments as per Boughton (1997).
V_P	41 million m^3/year	Average yearly rainfall from pluviostations on Hay, Sleeman and Denmark Rivers. 1995-1997 data
V_E	-49 million m^3/year	Pan evaporation figures from Hodgkin and Clark (1988). Annual average by month. Inlet evaporation factored by 0.77 × pan evaporation as per Boughton (1997).
V_G	0.07 million m^3/year	Yu, (1998)
V_R	-148 million m^3/year	$V_R = -V_Q-V_p-V_G+V_E$
V_{Inlet}	118 million m^3	From bathymetry

Salt balance

For Wilson Inlet it can be assumed that $S_Q \sim S_P \sim S_E \sim S_G \sim 0$. This leaves S_x and S_R to assign. $S_{x\text{-}in}$ is the salinity of the inflowing ocean water, so that $S_{x\text{-}in} = S_{ocean.}$ The LOICZ guidelines recommend that $S_{x\text{-}out}$ is treated as the average salinity of the system (for small and compositionally homogeneous systems) or the salinity near the boundary between the ocean and the system (large or heterogeneous systems). The latter definition is more appropriate to Wilson Inlet, as the system is generally stratified when the bar is opened so is certainly not a homogeneous system. Thus $S_{x\text{-}out} \approx S_{out}$, where S_{out} represents the average salinity of the outflowing water. S_R is the salinity of the outflowing water and so is the same as $S_{out.}$

Equation (11) thus simplifies to the following:

$$0 = \sum (S_R V_R + S_{ocean} V_x) - \sum (S_{X-out} V_X) \quad (12)$$

and allows V_x to be determined:

$$V_x = \frac{V_R S_R}{(S_{out} - S_{ocean})} \quad (13).$$

The average salinities required for equations (12) and (13) are given in Table 4.

Table 4. Average salinities used in budget

Quantity	Value (ppt)	Source
$S_Q \sim S_P \sim S_E \sim S_G \sim 0$	0	
S_{ocean}	35	
S_{out}	18	Average salinity of sites WI 30, 6, 35 and 7, in top 2 metres when the bar is opened. Data from 1995 - 1997. This represents the average salinity of the outflowing surface layer.
S_R	18	

The values in Table 3 and Table 4 can be used to obtain a value for the exchange flow V_x using equation (13):

V_x = 157 million m^3/year.

This steady-state value of V_x is close to the average of the exchange volumes calculated individually for each year in

Table 2.

The residence time within Wilson Inlet (the average length of time water stays in the system) can be calculated by:

$$\tau = \frac{V_{Inlet}}{(V_x + |V_R|)} \quad (14).$$

The residence time τ = 0.39 years = 142 days = 4.7 months.

Nutrient balance

Assuming a steady state for both dissolved inorganic nitrogen (DIN) and dissolved inorganic phosphorus (DIP) over an entire year, and that the nutrient concentration in evaporated water and the nutrient contribution from groundwater and rainwater are negligible, equation (12) can be simplified to:

$$\Delta Y = -V_R Y_R - V_Q Y_Q - V_x(Y_{ocean} - Y_{out}) \quad (15).$$

The values required for the application of this table are given in Table 5 below.

Table 5. Average nutrient values used in the mass balance

Quantity	NH_4-N (mg/L)	NO_X-N (mg/L)	FRP (mg/L)	DIN (mmol/m^3)	DIP (mmol/m^3)	Source
$V_P Y_P \sim V_G Y_G \sim 0$, $Y_E \sim 0$	0	0	0	0	0	
Y_Q	0.06	0.22	0.03	20	1	Average of concentrations from all gauging stations in Wilson catchment. 1995 - 1997 data
Y_{ocean}	0.0025	.0025	.0015	0.4	0.05	Readings at WI 2 when completely flushed with fresh seawater.
Y_{out}	0.023	0.050	0.003	5.2	0.09	Average surface concentrations at WI 30, 6, 35, 34, 7, 9, while the bar is open. 1995 - 1997 data. This represents the average concentrations of the outflowing surface layer
Y_R	0.023	0.050	0.003	5.2	0.09	

Thus using equation (11) and the values in the tables the following values for ΔY can be determined:

Δ DIP = -1.4 x 10^5 mol/yr = -4 tonnes/yr
Δ DIN = -1.6 x 10^6 mol/yr = -22 tonnes/yr

The negative values for both ΔDIP and ΔDIN indicate that within Wilson Inlet DIP and DIN are taken up and that the system is a net sink of dissolved nutrients. The magnitude of these non-conservative transformations indicates a significant retention of the dissolved nutrients that came in to the Inlet from the catchment.

Approximately 85% of DIP and 50% of DIN that came in from the catchment in a year was stored in the Inlet.

In order to compare between systems, these values can be normalised by the area of the Inlet (48 km^2) to give the following:

Δ DIP = -3 mmol/m^2/year
Δ DIN = -33 mmol/m^2/year

These levels are low when normalised by the area of the Inlet, and could almost be considered negligible when the errors in quantity estimation are included.

The levels are however significant when compared to the total inputs. Although the Inlet does not receive a very large load of nutrients compared to some systems, such as the Swan Canning Estuary (Kalnejais *et al*, 1999), it does store a considerable proportion of what does comes in.

4.2.3 Nutrient sources and sinks

Another common way of assessing the fate of nutrients within an estuary is to relate the concentration of nutrients and other non-conservative parameters to salinity, a conservative parameter. This method, as with the LOICZ model relies on the fact that the concentration of a particular species within an estuary will be a linear function of its concentration in the river and the ocean if dilution is the only process acting to change the concentration of the species. If other biogeochemical processes are occurring then the deviations from linearity indicate non-conservative behaviour (Biggs and Cronin, 1981), the ΔDIP and ΔDIN terms in the LOICZ model. The behaviour of hypothetical substances in an estuarine environment is shown in Figure 28.

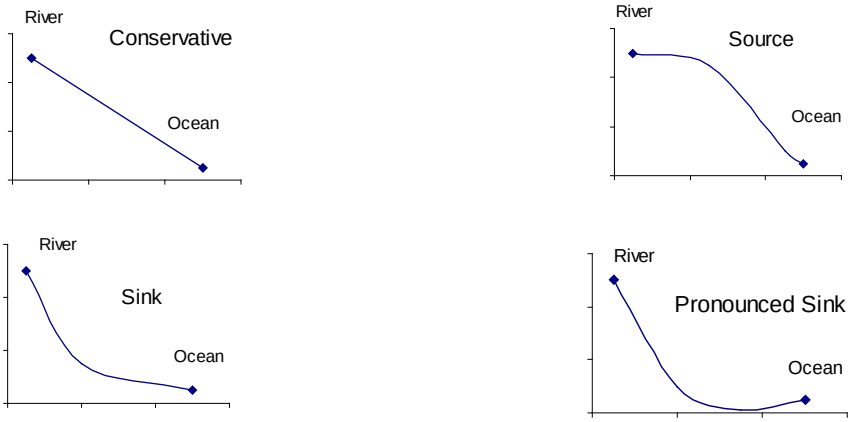


Figure 28. Idealised relationship between salinity and nutrient concentration, showing the mixing of nutrient-enriched river water and nutrient-poor ocean water. Adapted from Biggs and Cronin (1981)

The relationship between salinity and the concentration of important species in Wilson Inlet and its tributary inflows is shown in Figure 29. The interpretation of these plots is not necessarily simple due to the wide range of concentrations that came from the catchment at similar salinities and different discharge rates. The catchment data for dissolved species is only available from 1997, and is included just to provide an indication of the catchment input. Without a more detailed analysis and a more specific field programme the interpretations of these plots are limited to qualitative discussion. It is however evident that a linear relationship cannot be fitted to many of the species, indicating that many species undergo non-conservative behaviour.

Laboratory analysed colour demonstrated a relationship indicating a simple diluting effect. There was no colour data from the catchment, however although there is considerable scatter in the data a significant linear trend (r^2=0.62, P<0.001) between colour and salinity is evident. From a visual inspection of the data, the linear relationship is most distinct for salinities below 24 ppt.

Above this salinity the colour is lower than would be predicted by simple mixing between seawater and the coloured freshwater. This is possibly due to flocculation at the higher ionic strengths and subsequent settling out of the organic molecules contributing to the colour.

Total organic carbon concentrations measured in the Inlet demonstrated a linear relationship (r^2=0.36, P<0.001) between salinity, indicting that most of the TOC distribution pattern was due to dilution. There was however a considerable number of points deviating from the dilution line, suggesting that there may be non-conservative processes also influencing the TOC levels within the Inlet. The TOC levels from the catchment were all higher than the levels observed in the Inlet. This may indicate that there is a strong uptake of TOC within the Inlet, or may be due to inaccuracies in the analytical data between two laboratories.

The NOx plot is difficult to interpret because the large

range of values reported in the catchment make it difficult to determine if the high NOx levels in the Inlet were dilutions of highly concentrated catchment flows or if there was a source of NOx. However between a salinity of 24 to 26 ppt only low levels were detected in the water, lower than would be expected based on dilution of the 10 to 20 ppt Inlet waters. The Inlet was thus a sink for NOx at mid-salinities.

An internal source of ammonium, FRP and TP is evident between salinities of 20 and 32 ppt. At all other Inlet salinities the concentrations of these three species was low, indicting that there is likely to have been some uptake of these species (whether derived from internally recycled or external sources) within the Inlet.

The steady-state LOICZ budget indicates a net storage of both DIN and FRP (DIP). For this to be the case the sources of ammonium and FRP identified in the mixing plots must be of a smaller magnitude then the sink terms that are also evident.

The TN and silica plots are difficult to interpret due to the scatter of points. For both species there does seem to be some high levels in the 20-30 ppt range that are too high to be accounted for by dilution of the very high levels that were observed in the catchment, indicating an internal source.

Mixing plots and the LOICZ model have given an indication of some of the net processes occurring within the Inlet, and the salinities at which they occur. However they do not provide information into the dynamic processes acting in the Inlet. The following three sections deal with some of the processes responsible for the data distribution evident in the mixing plots.

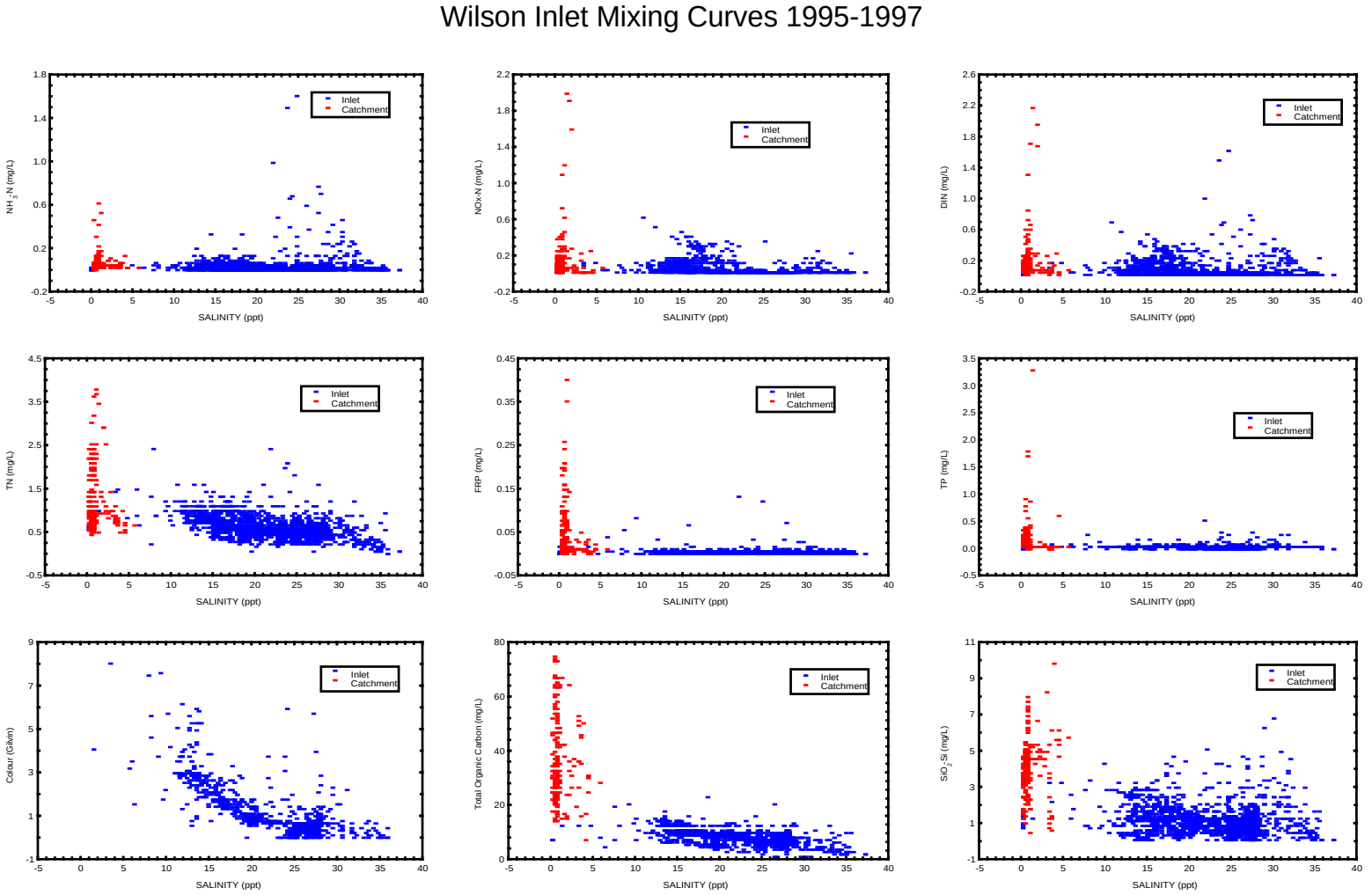


Figure 29. Mixing plots, showing the relationship between salinity and nutrients level in the catchment and inlet

4.2.4 Catchment derived species

The levels of NOx and TN in the Inlet can be seen to rise due to the winter riverine flow (Bastyan, 1996, Thompson, 1997). A rise over several weeks in the NOx and TN level was reported in each year after the onset of heavy river flow (Figure 31). In 1995 the first rise in levels was reported in August, while in 1996, the heavy rainfall was earlier and NOx and TN started to rise in July. The peak in NOx level in 1997 was lower than in the other two years and did not commence until September. The riverflow in the three years was quite different as shown in Table 7 and Figure 30, and the concentrations of NOx were higher in 1996, the wet year, than in the other two years.

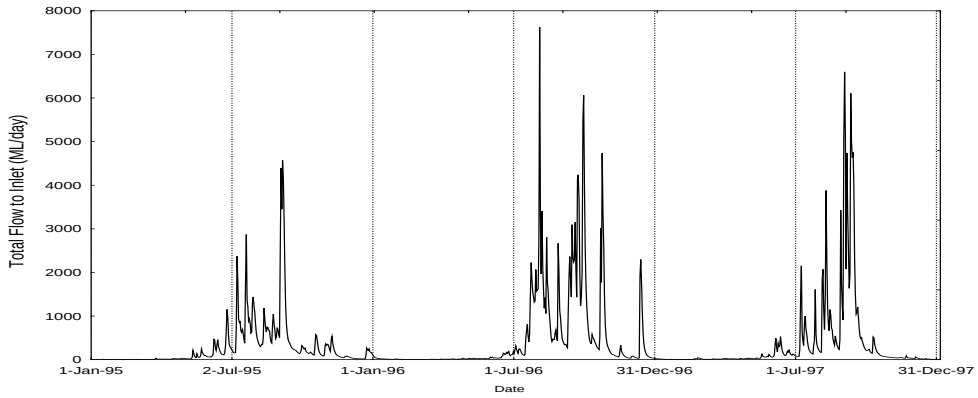


Figure 30. Total riverflow into Wilson Inlet.

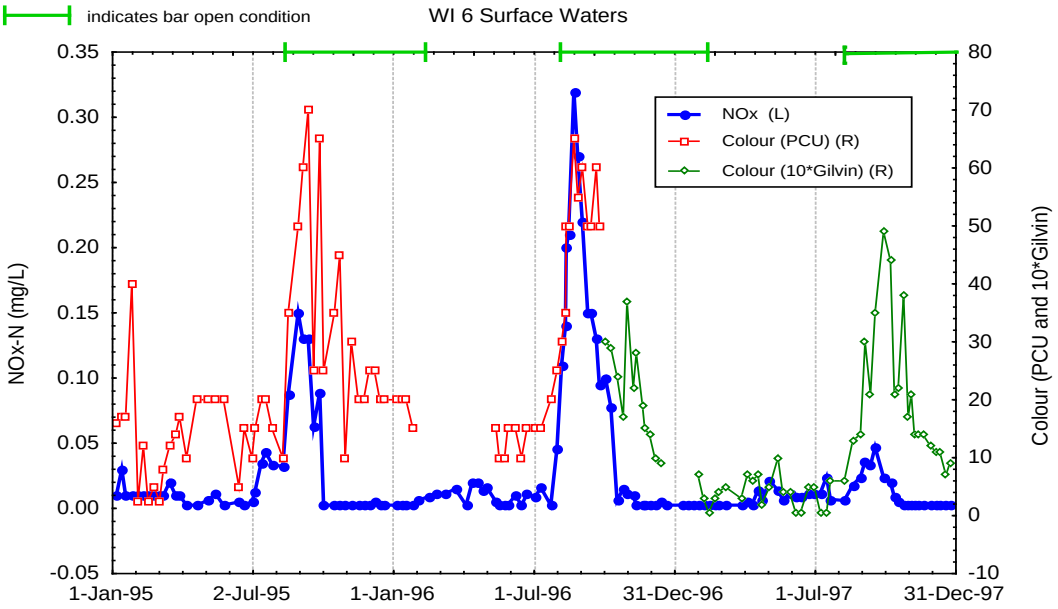


Figure 31. Concentrations of NOx and levels of colour in the surface waters of site WI 6

Note the change in colour parameter in 1996.

Table 6. Total flow volume (megalitres) recorded at the gauging stations on the four major inflows

Total Flow (megalitres)	Denmark	Hay	Cuppup	Sleeman	Total
1995	21 388	37 439	7 483	8 738	75 048
1996	44 501	71 708	9 278	13 128	138 615
1997	23 897	37 871	9 940	13 289	84 996

Site WI 14 and WI 12 generally recorded the highest NOx levels due to their proximity to the sources of nutrient inputs. WI 7, which is very close to the mouth of the Demark River reported lower values than WI 12 and WI 14 because the Denmark River was less loaded with nutrients than Cuppup, Sleeman and Hay. High NOx levels were reported at the other sites in the Inlet basins, coincident with the downstream movement of the freshwater plume. The intruding seawater was of lower NOx levels than the Inlet water as evident by the levels close to the detection limit of 0.05 mg/L that were reported at WI 2 when saltwater filled the whole water column. The saline bottom layer thus often reported much lower levels of NOx than the fresh surface waters above them. This was however not always the case, with NOx levels of up to 0.2 mg/L in the saline bottom layer reported on 10 occasions at sites ranging from WI 2 to WI 35. This may be due to the supply of NOx from denitrification in the sediments.

The catchment monitoring also demonstrated that TP and FRP were carried to the Inlet, particularly from the Cuppup and Sleeman Rivers. Yet, unlike NOx, no distinctive peak in either TP or FRP was evident. Only a few incidences of elevated FRP (between 0.02 and 0.08 mg/L) in the surface waters were reported. These high levels occurred at WI 7, WI 14 and WI 12, the sites closest to the riverine input, and never for more than one sampling run. The occurrence of the FRP in the surface waters can be correlated to high riverflow in the preceding days. In order to detect elevated levels of phosphorus in the Inlet, it seems that the sampling needs to be conducted close to the time when the P first reaches the Inlet. The general lack of phosphorus in the waters of the Inlet indicates that it is being removed from the system very quickly. The P is either being taken up by the organisms in the Inlet or is settling to the sediments. The *Ruppia* meadows close to the river mouths could be acting as important controls, slowing the flow of water so causing any particulates (and their adsorbed nutrients) to settle in their vicinity (Bastyan, 1996).

The elevated nitrate levels remained longer than any elevated phosphorus levels, but still did not persist for long in the Inlet, with the NOx peak dropping rapidly over September and reaching baseline levels by November (Figure 31). As the NOx level within the Inlet was high in August at the time of the bar opening, considerable NOx would have been flushed out to the ocean with the opening of the bar. The rapid decline in NOx which was observed was unlikely to be due solely to a flushing and diluting effect due to the outflow of freshwater and the inflow of marine water, because a similar rapid decline is not observed in other species. Colour for example is also a catchment derived species, which is not found in marine water. A rise in colour levels in the Inlet also occurred over July/August, but with a peak in September and then a gradual decline through summer, reaching low steady levels by February (Figure 31). The persistence of colour (which from the mixing plot analyses, behaves nearly as a conservative tracer) indicates that the flushing and diluting of the Inlet is not sufficient to explain the rapid drop in NOx. The removal of NOx must have been an internal process, and contributes to the ΔDIN sink term calculated in the LOICZ budget. The relationship between the water temperature and NOx level in Figure 32 shows that above 18^{o}C very little (less than or equal to the detection limit) NOx persists in the water column. This behaviour is probably due to the biological activity increase that is expected as the water temperature warms up after winter. With the increased activity the uptake of NOx increases, until the activity is at a high enough level to completely deplete the water of any NOx. The biological uptake of nutrients is discussed in more

detail in Section 4.2.6.

Ammonium levels in Cuppup creek were high, indicating the potential for some riverine input of ammonium. The other inflows however were low in ammonium, so the total ammonium input would not have been high. All sites did show a slight rise in the ammonium levels of the surface waters coinciding with the rise in NOx, but the surface ammonium levels still remained low. In each year the ammonium levels in the surface waters were reduced after the phytoplankton bloom. In 1995 however the ammonium level rose after the bloom and remained above around 0.05 mg/L for three weeks, before falling back down to low baseline levels. This ammonium peak coincides with a small peak in organic nitrogen and may be due to the breakdown of the organic matter in the dead phytoplankton cells.

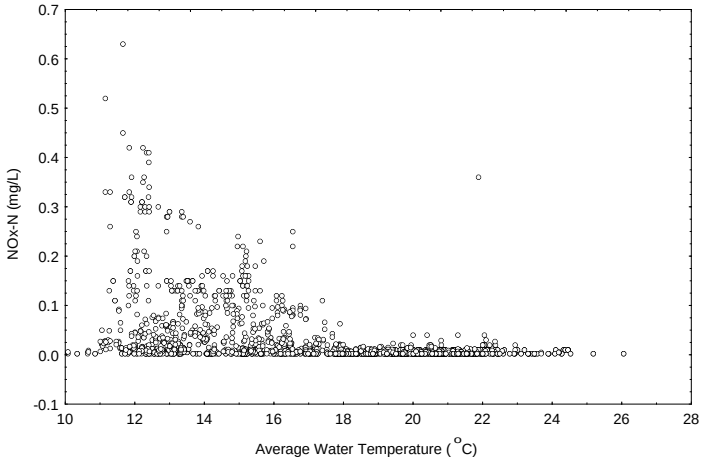


Figure 32. Relationship between concentration of NOx and water temperature for 1995 to 1997, all sites, all depths

A definite rise in silica occurred coincident with the rises in TN, NOx and colour, indicating that silica was carried into the Inlet with the winter river flow. The silica levels within the Inlet fluctuated more than any other species that was measured. From the annual high level associated with riverflow the silica levels declined, but not with the consistency with which NOx fell. Many small fluctuations with frequencies of two to three weeks were overlaid over a general decline in silica levels. Silica levels reached lowest levels in September, close to the time when the NOx levels had just declined down to their baseline level.

4.2.5 Internal nutrients sources

From the mixing graphs it is evident that there was a distinct internal source of ammonium, FRP, TP, TN and possibly silica, which occurred at a mid salinity range. The levels at this salinity range were the highest concentrations of these species that were observed, and from the seasonal plots it is evident that the internal sources produced these species between August and November.

4.2.5.1 Ammonium

Ammonium shows the most distinct evidence of an internal source. The high levels of ammonium occurred as peaks only in the bottom waters under stratified and low oxygen conditions (Figure 10 and Figure 33). The occurrence of the ammonium only in the bottom waters indicates strongly that the elevated ammonium concentrations are derived from the sediments. The ammonium peaks may be due to a number of individual or combined factors occurring in the sediments:

- pore water flushing due to the higher density seawater intruding into nutrient enriched porewaters (*pers. comm.* G. Douglas);
- simple restriction of vertical mixing by the stratification leading to the DIN released from the sediment essentially being concentrated in a smaller area;
- the low oxygen conditions limiting nitrification of the ammonium that is released from aerobic breakdown of organic matter in the sediments;
- anoxic breakdown of organic matter.

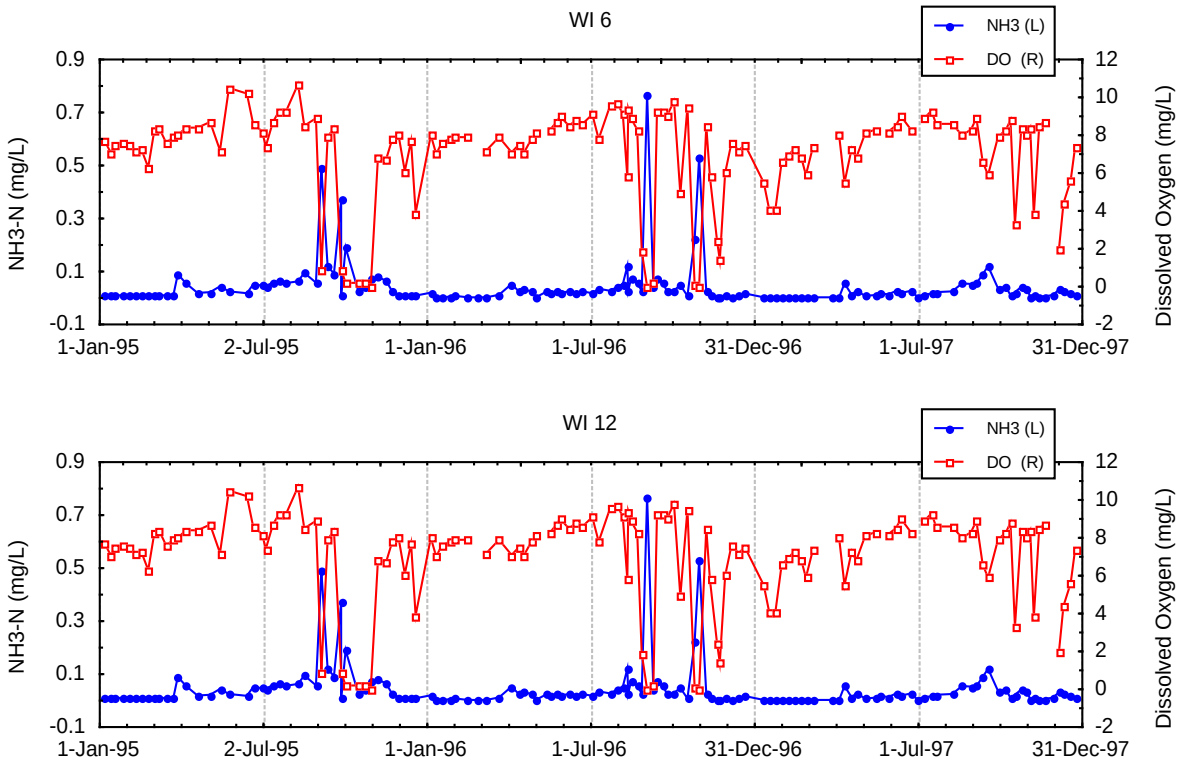


Figure 33. Relationship between water dissolved oxygen concentration and ammonium concentration in the bottom waters at sites WI 6 and WI 12

The relationship between oxygen level, duration of stratification and de-oxygenated conditions, ammonium concentration and TN concentration would be different for these mechanisms, so a more detailed examination of the temporal relationship of these variables should enable some of the more dominant mechanisms to be elucidated. With the weekly sampling frequency employed in the routine monitoring it is often difficult to categorically deduce which mechanism is appropriate.

Pore water flushing would be expected to occur after the first intrusion of water of a higher density. Figure 34 shows the relationship between ammonium concentrations in the bottom waters and the vertical density gradient. The density of the water was calculated from the salinity and temperature data using the UNESCO (1981) Equations of State. The vertical density difference was then defined as the density difference between the surface and bottom hydrolab samples. Thompson (1997) pointed out that high salinity water was present at WI 6 a month before high levels of ammonium were evident, and eliminates pore water flushing as an important mechanism. The lag time between a significant density stratification and nutrient release varied from between 2 to 4 weeks. Only WI 12 in 1995 and 1996 showed an elevated level of ammonium coincident with the first intrusion of saline water. Although these data do show little evidence for porewater flushing, with the weekly sampling frequency there was the possibility of missing a flushing event, particularly if the event was fairly rapid. More information on porewater flushing as a nutrient release mechanism will be obtained from the NEMP funded sediment study.

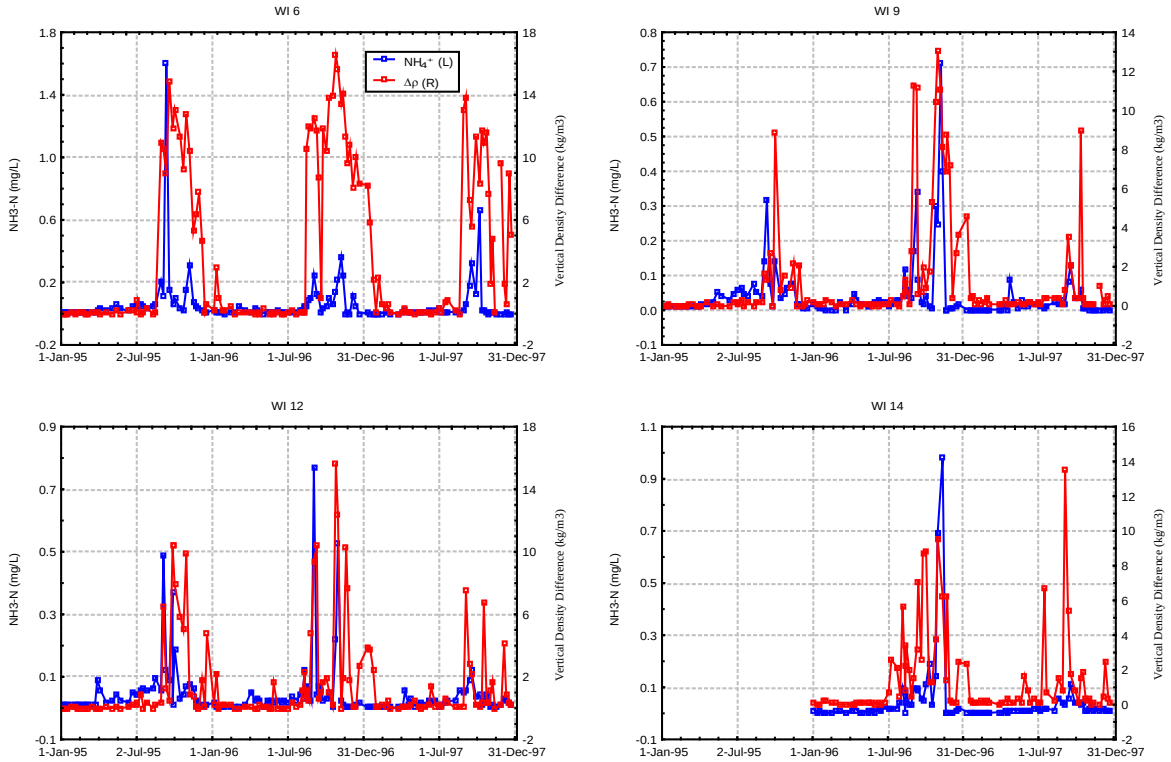


Figure 34. Ammonium concentrations and vertical density difference for the basin sites

The salinity stratification that is set up due to the intrusion of marine water can act as a barrier against efficient mixing. This barrier can partially isolate the saline bottom layer from the surface waters. Due to the restricted fluid motions in the bottom layer, molecular diffusion is the primary transport process that can move species into or out of the layer. As molecular diffusion is a far slower transport process than the mixing due to fluid motions, species within the bottom layer are partially confined and isolated within that layer. Any species derived from the sediment, which reach the water column under a stratified condition, are essentially concentrated within the bottom layer.

This concentration effect allows elevated levels of nutrients to be detected in the bottom waters. There would be nutrient species released from the sediments under well mixed conditions, but they would be rapidly mixed into the water column and diluted to such an extent that they are not detected.

As species that are released into the bottom layer are concentrated, other species that are consumed within the layer become depleted. This is particularly important for oxygen, which is consumed as part of the diagenesis of organic matter. Low oxygen conditions thus develop in the bottom layer, as already discussed in Section 1.1.

The low oxygen conditions that were associated with salinity stratification show a very strong correlation to the occurrence of high ammonium levels. All of the high ammonium levels (> 0.1 mg/L ammonium) at WI 6 occurred under stratified and deoxygenated conditions below 4 mg/L, and 70% were reported at dissolved oxygen levels below 1 mg/L. Similar statistics apply to the other basin sites, other than WI 7. Very high ammonium levels, of up to 1.6 mg/L were reported at WI 6 and WI 35 in the western basin and levels of up to 0.8 at WI 9 and WI 12. Thompson (1997) suggests that WI 6 and WI 35 are in the area which is the source of nutrients, however the values from WI 9 and WI 12 suggest that the eastern basin is also potentially a significant source.

The high levels of ammonium were associated with the low oxygen conditions because dissolved inorganic nitrogen released from the breakdown of organic matter is blocked from passing thorough the nitrification/denitrification pathway. Nitrifying bacteria are unable to convert ammonium to nitrate below approximately 1 mg/L. This blocking leads to the build up of ammonium. Denitrification (the anaerobic bacterial conversion of nitrate to nitrogen or nitrous

oxide gas) is also stopped, as the nitrate substrate for denitrifying bacteria is no longer available. The potentially important denitrification route for removing nitrogen from the Inlet is thus blocked by anoxic conditions.

4.2.5.2 Stoichiometric analysis

To assess the relationship between deoxygenation and sediment nutrient release the apparent stoichiometry of the reactions can be studied (*pers. comm* D. Heggie) as shown in Figure 33. The molar concentrations of a particular species are plotted against the molar oxygen deficit (ie oxygen level at 100% saturation - measured oxygen level). The predicted nutrient level at a given oxygen consumption is also plotted for organic matter with the Redfield composition of 106C:16N:1P (Redfield, 1958). This assumes that phytoplankton with the Redfield ratio are the principle forms of organic matter within Wilson Inlet. The stoichiometry assumed is the following:

$$(CH_2O)_{106}(NH_3)_{16}(H_3PO_4)+106O_2 \rightarrow 106CO_2 + 16\ NH_3 + H_3PO_4 + 106\ H_2O$$

and under aerobic conditions the ammonia will be nitrified to nitrate, giving the overall reaction.

$$(CH_2O)_{106}(NH_3)_{16}(H_3PO_4)+138O_2 \rightarrow 106CO_2 + 16\ HNO_3 + H_3PO_4 + 122\ H_2O.$$

For this analysis to give an accurate indication of the stoichiometry, a water parcel needs to be isolated from the rest of the water column. This is not the case with the monitoring data collected. The stratification is used as the isolating mechanism, and no indication is available to assess how effective it is. There will be an exchange of oxygen and nutrient species across the sediment/water interface and the halocline that will impact on the stoichiometric analysis. However as a preliminary assessment of sediment geochemical processes it is valuable to assess the information that can be generated.

From the plot of DIN and oxygen consumption it can be seen that about 20% of the DIN levels reported at very low oxygen concentrations fell above the predicted line. There are two explanations for these DIN levels greater than predicted from Redfield stoichiometry:

1. The oxygen deficit is an underestimate of oxygen consumption as a result of transport of oxygen from above the halocline into the bottom layer.

2. Instead of oxygen acting as an electron acceptor in the above reactions other sediment species such as iron, manganese, nitrate and sulphate may have reacted (Froelich et al., 1979).

Most of the high ammonium levels were reported in an anoxic water column, suggesting that sulphate reduction may be the important reaction, as the other three species react under sub-oxic conditions. The strong smell of hydrogen sulphide associated with the sediments and high levels of sulfide reported in Wilson Inlet sediments (*pers comm* D. Fredericks) adds further evidence to the occurrence of sulphate reduction of organic matter.

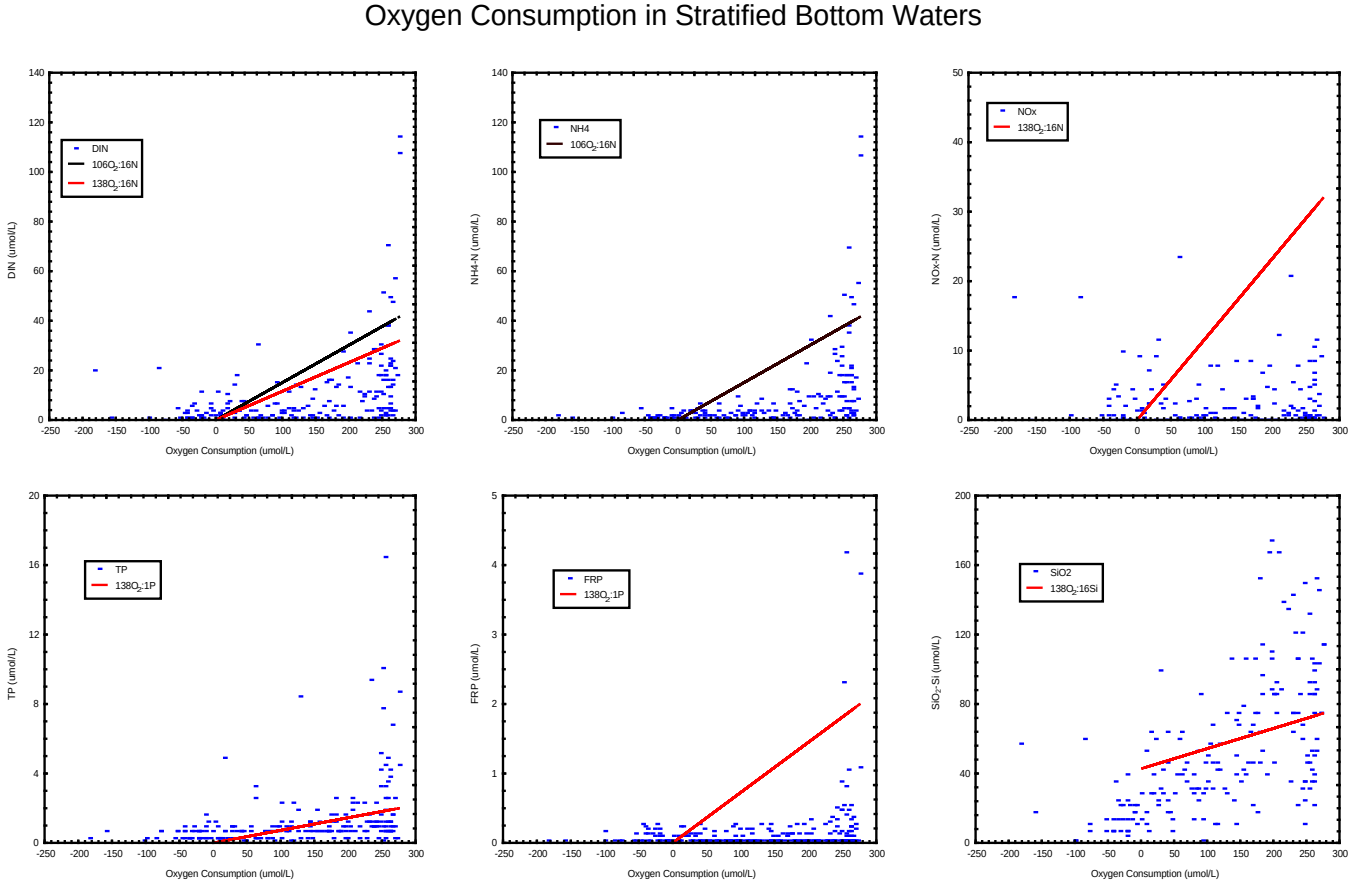


Figure 35. Oxygen consumption and nutrient concentration in bottom waters when stratified

The linear fit is that predicted from aerobic breakdown of organic matter with a Redfield ratio. Moving from 0 through to 300 µmol/L on the x-axis is increasing the extent of deoxygenation. Consumption of greater than 250 µmol/L represent water samples that are anoxic, or close to it. Negative values indicate water super-saturated with respect to oxygen, most likely due to photosynthesis.

4.2.5.3 Spatial extent of sediment nutrient release

The spatial extent of the ammonium release varies between sampling runs, and as expected was dependent on the extent of the deoxygenated conditions. On some runs high levels of ammonium were reported over large areas of the Inlet, from WI 30 to WI 12. This occurred on 5 and 12 September 1995, 2 September, 25 and 30 October 1996 and 17 September 1997. On other occasions only the sites in the western basin showed elevated levels, this generally occurred when only the western basin was stratified and the eastern basin was well-mixed. In order to determine the vertical extent of the elevated ammonium levels several additional samples were collected. These samples were generally collected just above the halocline, and revealed that the elevated levels only persisted in the saline layer, and not in the overlying water column.

4.2.5.4 Duration of elevated ammonium levels

The ammonium levels remained elevated for no more than four weeks at a time, and generally for only a single week. As suggested by the mixing plots there was a distinct internal sink for ammonium, ie internal recycling. The rapid disappearance of ammonium may indicate that it was taken up from the bottom waters by the biomass of the Inlet. This would suggest that predominantly benthic algae or phytoplankton with the ability to migrate to bottom waters were able to take advantage of the high levels of ammonium. However the fall in ammonium levels was probably more dependent on whether the bottom waters were refreshed either with new marine water or by mixing with the rest of the Inlet waters. Generally the ammonium peaks only lasted as long as the oxygen levels remained low. This would suggest that either the ammonium is mixed in with the Inlet water, where it would be available to more species, including the *Ruppia* and its epiphytes, or with the establishment of oxygenated conditions nitrification was able to start up again. It is also possible that the ammonium rich bottom water was moved away from the sampling sites by the fresher saline water, but still remained as a bottom layer, and so accessible only to the species mentioned above.

The maximum reported duration for a de-oxygenation event was 33 days (and could have been up to 40 days due to the weekly sampling regime), which occurred at WI 12 from 27 September 1995. This coincided with very still weather conditions. The ammonium level peaked at the same time the deoxygenated conditions were first reported, but declined within two weeks. The persistent low DO level may indicate that no refreshing saline water reached WI 12 over this time, so no physical mechanism, other than diffusion could transport the ammonium from the bottom layer. It seems likely that in this case low DO tolerant MPB or dinoflagellates would have consumed the ammonium. Of course the bottom water may have been refreshed during this time, but the DO levels had fallen below 1 mg/L by the time sampling was performed again. Data loggers deployed on the bottom of the Inlet, recording salinity and dissolved oxygen concentrations will provide further information on refreshment rates in the bottom waters.

In contrast, a low DO event at WI 9 which started on 11 November 1996 and seems to have lasted about 17 days reported high ammonium values over the entire period, with levels not returning to normal until the eastern basin became well mixed. The uptake and transformation of ammonium will be discussed in more detail in Section 4.2.6.

4.2.5.5 Phosphorus

Filterable reactive phosphorus occurred at elevated concentrations in the bottom waters only when elevated levels of ammonium were also reported. FRP levels of up to 0.13 mg/L were reported at all basin sites, both east and west. The molar ratios of ammonium-N to FRP-P varied in these nutrient enriched waters between 8 and 122, with no discernible tendency. As with ammonium, the FRP must be derived from an internal source (see Figure 27) namely the sediments and is released in greater amounts under anoxic conditions (Figure 33).

From Figure 27 it is evident that the number of points indicating an internal ammonium source were considerably greater in number than those contributing to FRP. As discussed by Thompson (1997) not all the peaks in ammonium were accompanied by FRP peaks.

Thompson analysed the water quality data by looking at Inlet wide averages and showed that on an Inlet wide scale two peaks in ammonium occurred in 1995, and these coincided with the FRP peaks. Then in 1996 two ammonium peaks again occurred, but there was no FRP peak coinciding with the first ammonium peak.

When the data is assessed on a site by site basis it is evident that elevated ammonium levels, without elevated FRP peaks were more common than a single event. In 1995 at sites WI 9 and WI 12 ammonium levels reached up to 0.5 mg/L, but no phosphorus peaks occurred. Also the second ammonium peak at WI 6 in 1995 was not accompanied by an FRP peak. The inlet wide averages produce a coinciding FRP peaks because high values of FRP (greater than 1 mg/L) were reported at WI 35. In 1996 only site WI 12 had a FRP peak to coincide with the first ammonium peak. In 1997 no FRP peaks coincided with the ammonium peaks at WI 9. The only site in which the ammonium peaks was always accompanied by a FRP peak was WI 35.

Potential reasons for this behaviour are:

- the FRP peak did not occur (Thompson, 1997);
- FRP release did occur but due to different chemical controls was removed from the water column more rapidly then the ammonium;
- the sampling regime missed a peak (Thompson, 1997);
- biota rapidly consumed the phosphate from the water column (Thompson, 1997);
- the sample preservation was inadequate due to the anoxic conditions and resulted in a transformation of phosphate (*pers comm*, A. Longmore) so it was not detected as FRP.

Different biogeochemical controls act on the release of ammonium and phosphate even though they may both be derived from the same organic matter. In the presence of suitable cations, such as iron (III) and manganese (III, IV) insoluble complexes are formed, which act to bind phosphate, so preventing its release. The ammonium peaks that are unaccompanied by FRP peaks may be due to trapping of the released phosphate. The phosphate peaks that are observed may indicate that the trapping mechanism had broken down. Under anoxic conditions the insoluble complexes capturing the phosphate may dissolve, so releasing additional phosphate to the water column. This would elevate the phosphate level to well above the predicted Redfield concentration. This mechanism may account for the few very high levels of FRP that were reported.

Based on the FRP peaks that are detected it is evident that phosphorus is very rapidly removed from the water column. Nearly all of the occurrences of non-coinciding FRP peaks are at WI 12 and WI 9. These are the shallowest of the sampling sites, and so would be expected to get higher light penetration to the bottom. Microphytobenthos (MPB) colonising the shallower sites would be perfectly positioned to intercept any phosphorus that is released from the sediments. The FRP peak was detected at WI 35 possibly because in winter the light penetration is poor (secchi disk readings less than 2 metres) and the MPB were unable to function and take up the FRP.

The final explanation for the missing FRP peaks could be that the FRP in the bottom waters underwent transformations once sampled. Although this is conceivable as the preservations routinely used are not specific for water sampled from anoxic conditions. However if there was a transformation of FRP within the sample bottle, then the phosphorus could still be picked up as TP. None of the samples with elevated ammonium and low FRP had high levels of TP, implying that this is unlikely to be the cause of the absent peaks.

The FRP peaks that were reported only lasted for one sampling run, with the FRP levels always completely depleted by the time the next sampling run was performed (ie within a week). As the elevated levels did not remain for more than a week it is quite possible that a weekly sampling regime could miss the FRP peak altogether. The disappearance may be due to biological uptake or chemical removal. The absence of FRP peaks is a strong indication that either sediment – phosphate interactions and/or biological interactions are very important for controlling the phosphate concentration in the water column.

4.2.5.6 Silica

The mixing plots showed evidence of an internal silica source. From the data in Figure 35 there was a significant relationship (r^2= 0.6, P<0.01) between oxygen consumption and silica level in the bottom waters. The levels of silica in the water are rarely below the detection limit, as is the case for many of the other dissolved nutrients. So it is incorrect to have the predicted silica level at zero oxygen consumption set at zero. For Figure 35 the predicted silica level at 0 oxygen consumption has been set at the average silica value for the months from June to September. This is to give an indication only. Because the levels of silica fluctuate more than any other species it is difficult to

assign such a value. The predicted silica release due to the breakdown of organic matter derived from diatoms (Webb, 1981) is also plotted, assuming the initial average concentration in the water.

The observed silica levels fluctuate about the predicted line. It is difficult to accurately interpret this data, as the initial silica concentration for each point is not known. A more specific study is required to account for the silica data.

4.2.6 Uptake of nutrients within the inlet

High nutrient levels occurred in the waters of the Inlet between July and December. The dissolved inorganic nutrients, inparticular NOx, NH4 and FRP, did not remain in the water column for extended periods, indicating that they were removed very quickly.

From the nutrient-oxygen plots, it is evident that at an oxygen consumption between 0 to 200 μmol/L, the levels of DIN and FRP are considerably lower than predicted. This may be due to several factors:

- the organic matter that is breaking down is not Redfield, (measurements of carbon release are required to determine this)
- the oxygen deficit is an overestimate due to transport of oxygen into the sediments.
- denitrification is removing the nitrate-nitrogen;
- biological uptake removed the nutrients;
- the nutrients reacted or adsorbed rapidly and so were not detected in the water column.

From the data available it is difficult to differentiate many of the above processes. The three NEMP funded projects on Wilson Inlet should provide more detail about the importance of the above processes. For the moment however it is possible to calculate some mass balances and to speculate on the fate of the nutrients in the water column.

4.2.6.1 Nutrient inventory

To assess the relative importance of the dissolved nutrients an inventory was calculated for each of the three years as shown. The values presented are the maximum 'equivalent Inlet wide nutrient concentrations' that were reported on a single sampling run. The equivalent Inlet wide concentration represents the concentration in the Inlet if it were to be well mixed. In the case of NOx, which was distributed uniformly throughout the Inlet, the concentration is the average (over the 7 sites and the two depths) NOx concentration. High levels of NH4 and FRP were generally reported only below the halocline and not throughout the whole Inlet. The depth of the nutrient enriched bottom layer and the interpolated extent over which this layer occurred was taken into account when calculating the equivalent Inlet wide concentrations of these species. The calculations are shown in Appendix B and the results are shown in Table 7 below. The conversion ratios for the biochemical composition of phytoplankton to equivalent levels of chlorophyll (Thompson, 1997) are also given as a rough indications of the expected chlorophyll level, if all the nutrient species were incorporated into phytoplankton under conditions in which nutrients were abundant.

Table 7. Maximum equivalent inlet wide nutrient and chlorophyll *a* levels reported in the Inlet prior to the peak of the phytoplankton bloom in each year

Species	Maximum Equivalent Inlet Wide Concentration (mg/L)			Ratio (g:g) Redfield (THOMPSON, 1997)	Equivalent chlorophyll (μg/L)		
Year	1995	1996	1997		1995	1996	1997
NOx	0.12	0.27	0.06	14N:1 chla	9	19	4
NH4	0.12	0.05	0.06	14N:1 chla	9	4	4
FRP	0.004	0.009	0.011	2P:1 chla	2	4	6
Silica	2.3	1.7	3.0	28Si:1 chla	80	60	110
Chlorophyll *a*	0.015	0.01	0.01	1:1 chla	15	10	10

4.2.6.2 Phytoplankton uptake

From Table 7 it is evident that the phytoplankton population in each year was capable of capturing most of the reported water column nutrients. Silica was the

only nutrient in excess in each year. This nutrient inventory is the maximum concentration for only a single day each year. Nutrients supplied by riverflow after the maximum concentration was reached would supply additional nutrients to phytoplankton, so that the 'maximum equivalent Inlet wide concentrations' and observed chlorophyll levels are not necessarily directly equivalent.

The maximum FRP levels reported in the water column would not have supported the chlorophyll concentrations observed in each year. Thompson (1997) did not find this to be the case for the 1996 data, and decided that the reported elevated nutrient levels in the bottom waters could have supported the phytoplankton growth. This is probably because the average Inlet concentrations calculated by Thompson (1997) did not reduce the weight of the high bottom water values to take into account the fact that the nutrient enriched bottom layer only occurred below the halocline (which was generally only 0.5 m thick). The Inlet average concentration calculated by Thompson (1997) is thus too high as it is a standard average of all values collected. In addition to this, the maximum Inlet average concentration of FRP used by Thomson (1997) was a value of 0.014 mg/L of FRP. This value occurred on 11 November 1996, at the same time that chlorophyll and phaeophytin values peaked. It is unrealistic to say that this 0.014 mg/L of FRP supported the chlorophyll bloom, as there was no subsequent bloom. The FRP peak of 11 November 1996 was most likely due to the mineralisation of dead phytoplankton cells.

The fact that the weekly sampling programme did not report sufficient FRP to support the observed chlorophyll *a* levels is probably due to the rapid removal of phosphorus from the water column. Either adsorption and precipitation or uptake by the Inlet biota, has resulted in the water quality sampling programme missing the high levels of phosphorus and hence underestimating the maximum Inlet wide levels of P in the water column. The discrepancy between chlorophyll and phosphorus indicates that phytoplankton uptake must be a significant sink for the water column phosphorus.

The equivalent Inlet wide concentrations of the nitrogen species show considerable variation in each year. In 1995 and 1997 there were approximately equal loads of ammonium and NOx, while in 1996, NOx was far in excess. In 1995 and 1997 for the levels of chlorophyll to have been supported by the reported levels of N species, the phytoplankton must have utilised both ammonium and NOx. This would indicate that the phytoplankton utilised nutrients derived from both the catchment and from sediment release. However, it is again difficult to make such statements as the sampling regime may have underestimated the maximum levels of these species (particularly of ammonium) due to their potentially rapid removal from the water column.

It is evident from this rough inventory that the relative importance to phytoplankton abundance of nutrients released from the sediments versus catchment derived species, is difficult to determine with the current sampling regime. This is due to only a basic knowledge of the extent of the sediment nutrient release (and hence accurate estimate of sediment derived nutrient loads) and the present lack of an accurate estimate for the catchment input of dissolved species (this data will soon be available).

To more fully understand the nutrient dynamics and improve the inventory estimates additional studies are obviously required. The water quality monitoring programme requires adaptation, so that the important events in Wilson Inlet are more adequately captured. For a better estimate of nutrient input due to anoxia it is recommended that samples be collected at additional depths and sites during severe stratification events. To assess any immediate effects of river inflow reaching the Inlet additional sampling in the shallow margins of the Inlet would also beneficial. This would assist in determining if significant quantities of nutrients are settling out or are rapidly taken up in the vicinity of the shallows and *Ruppia* meadows.

The temporal relationships between NOx, NH4 and chlorophyll may provide additional information on the importance of various species. The relationship between these three species is explored in Figure 25 and Table 8. It is evident that if the phytoplankton had responded solely to the input of ammonium, then there must have been a faster response time then if the phytoplankton were also utilising the NOx available in the water column. For the phytoplankton growth to have responded only to the ammonium, a fast response time due to warm, sunny conditions must have occurred (Thompson, 1997).

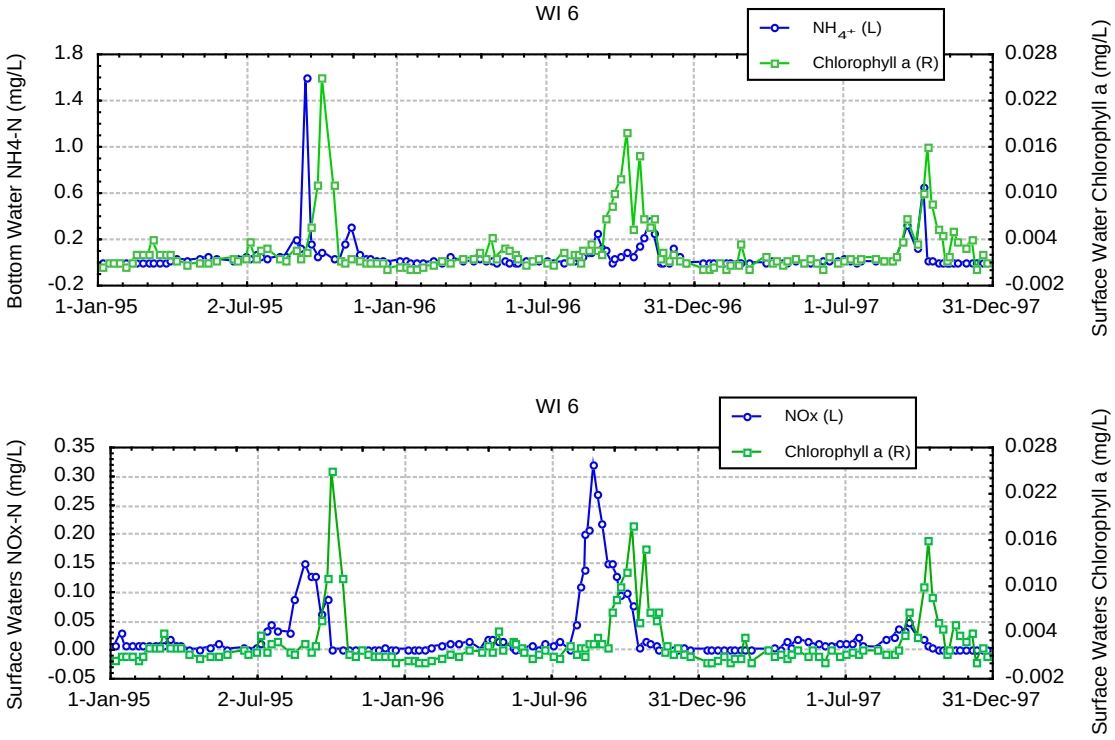


Figure 36. Time series of NOx, ammonium and chlorophyll data for site WI 6

Table 8. Dates of several important nutrient related events

Site	Year	Maximum NOx Level	NH4 bottom peak	Chlorophyll surface peak	Chlorophyll bottom peak	2nd NH4 peak	NOx Lag Time ± 14 days	NH4 Lag Time	Date Average Temp>18ºC
WI 6	1995	30-Aug	12-Sep	2-Oct	16-Oct	7-Nov	33	20	30-Oct
WI 6	1996	22-Aug	2-Sep	24-Sep	9-Oct	7-Oct	33	22	30-Oct
WI 6	1997	17-Sept	11-Sep	12-Oct	17-Oct	17-Sep	25	31	24-Oct
WI 12	1995	30-Aug	5-Sep	27-Sep	27-Sep	2-Oct	28	22	30-Oct
WI 12	1996	12- Aug	2-Sep	9-Oct	11-Nov	30-Oct	58	37	30-Oct
WI 12	1997	17-Sept	17-Sep	17-Oct	7-Nov	-	30	30	24-Oct

It is interesting that the chlorophyll levels in 1997 were similar to the previous two years, despite lower NOx and ammonium levels. This may indicate that there is some other control other than dissolved nitrogen concentrations on the phytoplankton ecology. The delivery of catchment derived nutrients to the Inlet was more gradual and later in 1997 due to the rainfall pattern. The timing and delivery of nutrient sources may also be important factors influencing the phytoplankton populations.

It is difficult to determine the relative importance of NOx, ammonium and other parameters on the phytoplankton populations from this data set. Additional research needs to be undertaken to determine the relative importance of each N species. The project 'Phytoplankton Ecology in Wilson Inlet' funded by NEMP should provide more information on the importance of each N species.

4.2.6.3 Nutrient uptake by Ruppia and other primary producers

Phytoplankton are not the dominant biomass in the Inlet. *Ruppia* is the largest biomass and epiphytes and benthic algae are other important components of the

system. If phytoplankton removed most of the nutrients from the water column between August and October then these other species must have had an alternative nutrient source.

It is possible that these species (together with the phytoplankton) rapidly removed nutrients from the water column and so utilised water column nutrients that were not detected by the sampling programme. If this is the case, due to the greater spatial extent of NOx, it may be the more important N species for *Ruppia* and its epiphytes, while phytoplankton able to reach the elevated levels of ammonium and FRP below the halocline may use more ammonium.

There is an additional source of P, other than that measured in the water column. The tissue phosphorus concentrations of *Ruppia* are related to sediment nutrient levels, suggesting that root uptake of phosphorus is important (Lukatelich *et al.*, 1987). Benthic algae due to their location are also likely to be able to use sediment nutrient sources.

The impact of Ruppia on the water quality may differ each year. The Ruppia biomass is not stable, so that its importance as a nutrient sink may vary each year (Carruthers, 1997). Also considering the large variations in nutrient inputs from the catchment and sediment release that have been observed over 1995 to 1997, the importance of various species in nutrient uptake is likely to differ in each year.

4.2.7 Post phytoplankton bloom summer period

Following the collapse of the phytoplankton bloom a second ammonium and FRP peak occurred in each year at most of the bottom waters of the basin sites. The timing of this peak when compared to the phytoplankton bloom varied between site and year. It is likely that the ammonium and FRP levels were due to the breakdown of dead phytoplankton cells.

After November of each year there was very little nutrient or phytoplankton activity for the remainder of the dry season. No successive phytoplankton blooms occurred after the second ammonium release. The second nutrient peaks indicate that at least some of the nutrients incorporated into the phytoplankton were recycled into the water column. The levels of nutrients in the bottom water at this time, once mixed into the rest of the Inlet were however insufficient to support another significant phytoplankton bloom.

The decaying phytoplankton bloom supplied the sediments with new organic material. The size of the nutrient release following the bloom does not account for the full magnitude of the bloom observed. Thus there was storage of organic matter in the sediments. This material, unless buried would be available for future diagenesis and hence nutrient release to the water column.

The TN levels in the water column fell to the lowest levels of the year in January. Based on the normal biochemical composition of phytoplankton (Thompson, 1997) that 14 mg/L nitrogen is converted to 1 mg/L chlorophyll *a*, the reduction in the surface chlorophyll level in the water, for example by 0.024 mg/L (WI 6, 1995) would lead to a 0.35 mg/L reduction in the TN. The observed reduction in TN is 0.5 mg/L, suggesting that a considerable proportion of the fall in TN over October to December would be due to the phytoplankton bloom decline. As the TN levels do not become elevated again until the start of the rainfall in the following year, much of the nitrogen that was in the phytoplankton did not reappear in the water column.

By December the TN values were steady, reaching a median value of 0.44 mg/L, which persisted until March. Another smaller peak in organic nitrogen occurred in April/May, with a median of 0.8 mg/L. This secondary peak occurred when the salinity in the Inlet was highest and turbidity was at a minimum. No increase in phytoplankton cell counts was reported during this time so it is likely that other organisms were contributing to the organic nitrogen increase. The epiphytic macroalgae biomass on *Ruppia* is higher in April than in June (Carruthers, 1997) and epiphytic marine phytoplankton species would be expected to find the conditions in April more conducive to higher growth rates, than earlier when the salinities were lower. The diatoms produce matrices of mucus, diatoms and other particulate matter (Carruthers, 1997), which may slough off from their substrate and contribute to the organic nitrogen found in the water column. Senescing *Ruppia* may also have contributed to the increase in TN.

Following the winter peaks of NOx, NH4, FRP and TP baseline levels of these species were maintained until significant riverflow or stratification occurred in the

following year. Once the intrusions of marine water had ceased, due to the shoaling and closure of the channel to the ocean, stratification events no longer occurred, the Inlet was well mixed so that the oxygen levels remained high. There were thus no large scale anoxic sediment nutrient releases, and any nutrient realised from the sediments must have been rapidly utilised as no dissolved nutrients were detected in the water column. Biological uptake or denitrification must have been acting to keep the water column free of bioavailable nitrogen.

4.3 Comparison with historical data

It is extremely valuable to compare the water quality data collected in the Inlet over 1995 to 1997 to the other historical data sets that are available. By doing this it is hoped that any degradation or improvement in the water quality in the Inlet can be quantified. Unfortunately it is often difficult to make definitive statements about changes in water quality because of the differences in sampling programmes and analyses used in the past.

Lukatelich *et al.* (1984) compared their sampling data with that of Spencer (1952) by comparing 3 data points from the 1982/83 study with 13 points from 1945-1952. They found that each data set had a similar range and concluded that there was no evidence of an increase in concentration from the 1950's to the 1980's. A comparison of the Lukatelich *et al.* (1984) data, which is the most comprehensive historical data set available, and the data from this study is given in Table 9. The sample collection methods used for each study are comparable, with the surface water collected in the same manner. The bottom sampling methods were different, but would have sampled from a similar depth from the bottom. The Lukatelich *et al.* (1984) study employed a diver to sample 10 cm from the bottom, while this study sampled from the depth just above the bottom that did not cause the hose sampler to entrain sediments. The 1995 to 1997 samples may have been collected a couple of centimetres closer to the sediments than the previous study.

The Lukatelich *et al.* (1984) report did not provide standard deviations or other statistical measures of the data, so a rigorous statistical analysis of the two data sets is not readily performed. The considerable difference in sample size and sampling frequency also make identification of any changes in water quality difficult. The 1995 to 1997 summer averages for most parameters are lower than in 1982/83. This may be due to a greater biomass taking up nutrients after winter and so depleting the water columns of nutrients. Although it is generally accepted that the biomass of Ruppia was greater in the 1980's there may have been lower total biomass as the epiphyte or MPB populations may have increased.

The averages in winter of surface ammonium, FRP and [TP-FRP] are less or comparable, possibly indicating that there is now less catchment input of these species. Alternatively there may now be increased uptake of these species. The current levels of NOx are considerably greater, 1982/83 was a dry year and as NOx is catchment derived, less input in a dry year would be expected. The difference is quite pronounced however, so it would be prudent to assume that the NOx levels in winter have increased since the 1980's, this implies that more NOx is coming into the Inlet from the catchment.

The maximum values reported in both the surface and bottom waters in winter are all considerably higher than those in the 1982/83 study. The mean bottom values of ammonium and P species are also much greater. All of the high levels of these parameters reported in this study occurred only several times each winter and did not persist long in the water column, certainly for less than 4 weeks. Thus the greater (weekly) frequency of sampling over three years in this study would be expected to pick up a greater number of these events than the 6 weekly sampling that was performed in 1982/83. The maximum values are thus not directly comparable, as the sampling regime of the 1982/83 may have missed the events leading to the high levels. The slightly deeper bottom sampling depth employed in this study may also have lead to higher concentrations being reported. Again the maximum values are all much higher than those previously reported, so it is again prudent to assume that the maximum values observed in the Inlet have risen since 1982/83.

Table 9. The mean and maximum nutrient levels from Lukatelich et al (1984) and the present 1995 to 1997 study, all values in (ug/L). Errors in means are the 95% confidence intervals

		Summer				Winter			
		1982/83		1995-97		1982/83		1995-1997	
		Mean	Maximum	Mean	Maximum	Mean	Maximum	Mean	Maximum
NH4-N	S	27	57	11 ±1	80	23	27	25 ±2	110
	B	34	69	13 ±1	130	21	27	63 ±12	1 600
NOx-N	S	5	9	6 ±1	86	9	9	53 ±7	630
	B	6	10	6 ±0.5	40	8	8	43 ±6	450
TN	S			510 ±13	1 300			691 ±23	2 400
	B			507 ±16	1 200			625 ±25	2 400
TN-DIN	S	916	1120	492 ±13	1 293	934	1337	612 ±19	2 280
	B	926	1180	487 ±15	1 193	986	1382	519 ±19	1 698
FRP	S	2	4	3 ±0.2	12	4	5	3 ±0.5	82
	B	3	6	3 ±0.2	14	3	5	4 ±1	130
TP	S			15 ±1	70			23 ±2	270
	B			17 ±2	260			28 ±3	510
TP-FRP	S	50	88	12 ±1	69	37	41	20 ±1	216
	B	58	97	15 ±2	258	41	43	25 ±3	380
Chlorophyll a	S	2.5	4.7	1 ±0.1	7.4	3.6	4.9	3 ±0.4	65
	B	8.6	16.8	2 ±0.2	31	8.1	10.8	3 ±0.4	44
TOC				7500 ±180	17 000			9150 ±230	20 000
				7200 ±210	20 000			8050 ±230	23 000
SiO_2-Si				1110 ±100	5 000			1120 ±60	4 300
				1220 ±110	5 000			1240 ±70	4 700

The data collected by the Waterways Commission and Mr Owen McIntosh from the early 1990's can also be compared to this data set. A plot of the two data sets is given in Figure 37 and demonstrates the difficulties associated with comparing different data sets. As with the 1982/83 data the Waterways Commission data did not detect the occurrence of the very high levels of nutrients.

It is very difficult to say that the events that have lead to the very high levels in the water column started to occur after 1992. If this was the case it would mean that the factors contributing to the high values of P and ammonium, ie the anoxic events in the bottom water, are only just recent processes. It would also indicate that high levels of catchment derived species, NOx and TN are also recent occurrences. This seems unlikely as the catchments have been cleared for several decades and saltwater intrusions leading to stratification and deoxygenated conditions must have occurred after most bar openings.

The occurrence of peaks in high levels of nutrients in itself does not necessarily indicate a degradation of water quality, as the processes which lead to these events are highly dependent on the seasonal variations in rainfall, tidal levels and wind mixing events. What may indicate a degradation of the system is if there is an increase in the number of events each year leading to the release of high levels of nutrients, or if the levels of nutrients persist for longer in the water column (both measured over several years to remove any seasonal effects). In order to identify such changes it is necessary to maintain a sampling frequency that is able to detect such events. This means that a weekly sampling frequency at least, should be continued in Wilson Inlet during winter.

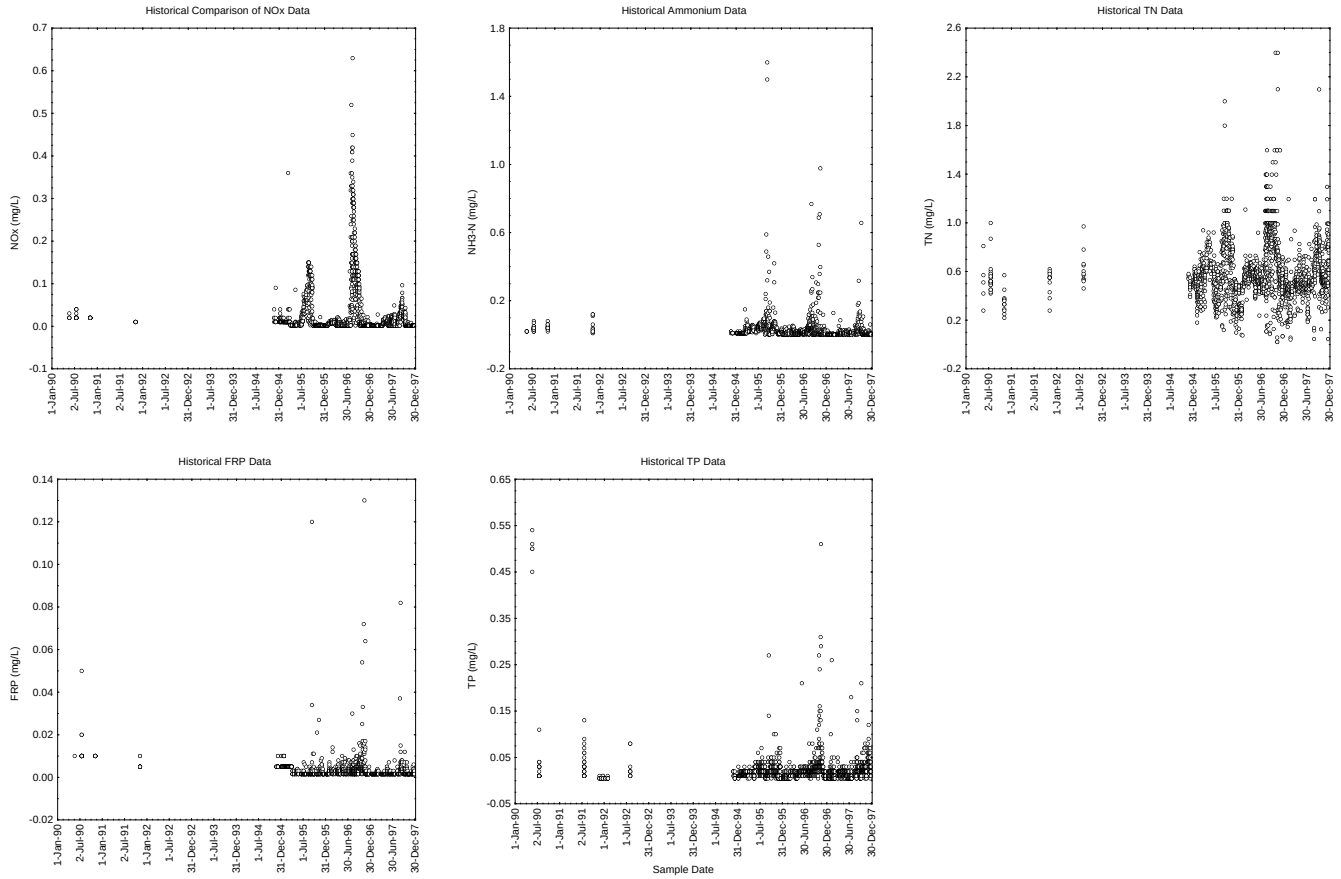


Figure 37. Time series plot of both Waterways Commission 1990 to 1992 data and the current study's 1995 to 1997 data

4.4 A nutrient enriched inlet?

For the years 1995 and 1997 the concentration of dissolved nutrients within the Inlet for over half the year was low and close to laboratory detection limits. Evidence of nutrient enrichment during this bar closed, dry period is thus hard to detect, with clear waters (secchi disk on the bottom) and very little phytoplankton activity. The levels of nutrients during this period were comparable or lower then the levels reported in the 1950's.

Considerable nutrients do however reach the Inlet, the catchment data indicates that high levels of both TN and TP, considerably above the ANZECC Guidelines are measured in the inflows. It is quite surprising that the Inlet does not show more signs of nutrient enrichment. Several important factors act to maintain the water quality in the Inlet.

The salinity stratification that is setup due to the marine water intrusion leads to deoxygenation in the waters below the halocline. The fact that anoxic events occur in Wilson Inlet, which is a shallow waterbody indicates in itself to Thompson (1997) that the Inlet is moderately eutrophic. The high levels of nutrients that were released from the sediments were only reported when the Inlet was stratified. Wind mixing is an extremely important process acting to breakdown the stratification and transport oxygen to the sediment water interface. The strong winds that often buffet the southcoast act to limit the length of time that severe stratification persists. The winds may thus be a factor moderating the eutrophic status of Wilson Inlet.

Although phytoplankton blooms occur only once a year, other excessive plant growth within the Inlet is evident. Over the past decade there were reports of excessive growth of the seagrass *Ruppia megarcarpa* within the Inlet (Lukatelich *et al*, 1982, Hodgkin and Clark, 1988). The *Ruppia* and its epiphytes, had a biomass far greater than the load of nutrients that came in from the catchments (Lukatelich *et al*, 1987). Combined with the presence of benthic algae, the primary producers in Wilson Inlet must be responsible for utilising much of the nutrients that come into the Inlet and hence keep the level of nutrients in the water column low for much of the year (Lukatelich *et al*, 1984; Thompson, 1997). The Inlet biomass is thus a significant store of nutrients. Denitrification may also be an important unquantified process removing nitrogen from the water column.

That the Inlet is storing a significant portion of the nutrients that come in from the catchment is a potentially dangerous situation. Should the anoxic events that have been identified in the last three years increase in spatial extent or duration (due perhaps to very still conditions, or an increase in organic loading from the catchment), then the internal store of nutrients in the organic matter of the sediments could be released to the water column in far greater levels then is currently observed. Although much of the nutrient released could be internally recycled back into the *Ruppia*, faster growing species of phytoplankton would be favoured to rapidly utilise the nutrients. This could lead to a shift in species composition and head down the path to more severe eutrophication. As Thompson (1997) says, "The Inlet is in a precarious balance".

5. Conclusions and future work

The Inlet has a highly seasonal hydrodynamic cycle. After the closure of the sand bar in January/February, the Inlet was well mixed, with uniform salinity, oxygen and temperature throughout the Inlet. In the closed dry state the salinity in the Inlet rose due to evaporation. The first freshwater inflow to the Inlet was mixed rapidly into the rest of the Inlet, resulting in an Inlet wide lowering of salinity. Freshwater plumes in the eastern basin were able to form after there was sufficient runoff and moved downstream as a buoyant overflow. With the mechanical opening of the bar, saltwater was seen to intrude into the Inlet within four days. The intrusion moved into the western basin, forming a saline bottom layer up to 1 metre in depth. Up to two weeks later the saltwater intrusion penetrated into the eastern basin, where the saline layer reached a thickness of 20 cm.

The salinity stratification that was set up due to the saltwater intrusion led to deoxygenation of the waters below the halocline, and anoxic conditions were reported on several occasions. The stratification was destroyed by strong wind mixing events, however the stratification re-established rapidly. As the channel through the bar shoaled, the saltwater entering the Inlet was reduced and the strength of the stratification reduced. Once the sand bar had closed the stratification was rapidly destroyed and the well mixed condition established.

The nutrient levels within the Inlet were closely related to hydrodynamic state of the Inlet. When the Inlet was well mixed the nutrient levels in the water column were low. With the inflow of water from the catchment, the nutrients species, NOx, silica and TN were carried into the Inlet. Levels of these species thus rose with the winter runoff. Some FRP was also carried in by the inflows. When the Inlet became salinity stratified, low oxygen conditions often became established below the halocline. When this occurred high levels of ammonium, FRP and silica were released from the sediments.

The concentrations of NOx, ammonium and FRP in the water column fell far quicker than can be accounted for by dilution alone. The Inlet was thus a sink for these species.

Phytoplankton levels in the Inlet reached bloom proportions once in each year over 1995 to 1997. The timing of the bloom occurred shortly after the peak in NOx levels and the sediment nutrient release. An inventory of water column nutrient levels indicated that the phytoplankton bloom could have captured most of the dissolved inorganic nitrogen and phosphorus species reported in the water column over winter. It is unclear from this data set which nutrients were the most important in controlling phytoplankton growth.

Following the collapse of the phytoplankton bloom, dissolved nutrients were briefly detected in the water column as the cells broke down. However after this, the nutrient level in the water column remained low through summer and until the winter rains. Any nutrients released to the water column over this period must have been taken up by Ruppia seagrass and its epiphytes and benthic algae. Denitrification may also be an important process keeping the nutrient concentrations in the water column low.

Considerable future work is required to understand the processes that are influencing the water quality within the Inlet. Additional water quality monitoring is required to determine the spatial and temporal extent of anoxic events and the associated high levels of ammonium and FRP.

The 1995 to 1997 data were collected in years with average or below average rainfall. 1995 and 1997 were both lower than the average rainfall, while 1996 was close to the average. Data for wetter years are also needed so that the Inlet conditions can be described for a full range of seasonal variations.

Biological uptake within the Inlet seems very important at removing nutrients from the water column. The species involved are most likely those with the largest biomass, namely phytoplankton and the seagrass *Ruppia*. The relative importance of each species is unknown. As these species could be largely controlling the nutrient levels in the Inlet it is essential to understand more about their nutrient uptake and

cycling.

Nutrients, in particular phosphorus and ammonium, are being released from the sediments. Salinity stratification leading to de-oxygenation of the deeper waters seems to be an important factor leading to high nutrient levels, however the mechanisms by which this is happening and the duration over which the releases occur are unknown. As the sediments are a potentially very large source of nutrients more detailed understanding of the processes leading to sediment nutrient release is necessary.

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Appendix A

Details of sampling programme

Sample regime

Weekly sampling at the following sites:

1995
physical data - WI 2, WI 30, WI 6, WI 7, WI 9, WI 12 and WI 14
chemistry data - WI 2, WI 30, WI 6, WI 7, WI 9 and WI 12
1996 -August 1997
physical data - WI 2, WI 30, WI 6, WI 7, WI 9, WI 12 and WI 14
chemistry data - WI 2, WI 30, WI 6, WI 7, WI 9, WI 12 and WI 14
September 1997 -May 1998
physical data - WI 2, WI 30, WI 6, WI 35, WI 7, WI 9, WI 12 and WI 14
chemistry data - WI 30, WI 6, WI 35, WI 7, WI 9, WI 12 and WI 14

Chemistry parameter suites and laboratory methods

Parameter	Laboratory	Laboratory Method Code	Method Description	Detection Limit (mg/L)
8/12/1994-30/4/1995				
NH4	CCWA	iAMMN1WAAA		0.02
NO3	CCWA	iNTAN1WAAA		0.02
FRP	CCWA	iP1WTAA		0.01
TP	CCWA	iPP1WTCO	persulphate total	0.01
TN	CCWA	iNP1WTAA	persulphate total	0.5
Silicate	CCWA	iMET1WCIP	filtered	10
Colour	CCWA	not specified		5
Chlorophyll	CCWA	iCHLA1WACO		0.001
Phaeophytin	CCWA	iCHLA1WACO		0.001
Beginning 1/05/1995				
NH4	AEL	PEW 063	Auto-Analyser (Phenate)	0.005
NO3	AEL	PEW 061	Auto-Analyser (Cd/Reduction)	0.005
FRP	AEL	PEW 064	Auto-Analyser(Ascorbic Acid)	0.003
TP	AEL	PEW 066	Digest/Auto Analyser	0.01
TN	AEL			0.05
Silicate	AEL	APHA 1995 4500 Si	Molybdateyellow/Visible spectrometry	0.1
TOC	AEL	MEI 040	Combustion, Infra Red	0.1
Colour	AEL	PEW 004	Visual Comparison	5
Chlorophyll	AEL	APHA 1992 10200 H	Acetone Extraction, visible spectrophotometry	0.0005
Phaeophytin	AEL		Acetone Extraction, visible spectrophotometry	0.0005
Kjeldahl Nitrogen	AEL	PEW 065	Digest Auto Analyser	0.05

CCWA – Chemistry Centre of Western Australia AEL – Australian Environmental Laboratories

Alterations and Additions to Sampling Programme				
15/06/96 –1/10/96 Colour	AEL	Colour Comparator	APHA 1995 2120B	5
Parameter	**Laboratory**	**Laboratory Method Code**	**Method Description**	**Detection Limit (mg/L)**
2/10/96 Gilvin	AEL	Filtered sample visible spectrophotometry at 440 nm	PEI 004	0.1
19/11/1996-17/4/1997 Selenium 12/8/96-27/8/96 no TOC	AEL	Hydride	PEW 025	0.002

Physical Data Parameter Suite and Instruments

Hydrolab H20 Multiprobe Specifications

Parameter	Units	Accuracy
Salinity	ppt	± 0.15
Temperature	^{o}C	± 0.05
Dissolved Oxygen	mg/L	± 0.2
Percentage Saturation	%	
Electrical Conductivity	mS/cm	± 0.01

Secchi Disk – 30cm diameter disk with black and white quadrants.

Site Locations

Abbreviation	Description	Easting	Northing
WI 12	Central East Basin	540 450	6126 900
WI 14	Southwest of Bream Rocks	542 400	6126 400
WI 2	South of Poddyshot Bay	530 267	6125 269
WI 3	Southwest of Poison Point	532 058	6126 299
WI 30	off Poddy Point	530 840	6126 087
WI 34	Central West Basin	534 648	6128 246
WI 35	Central West Basin	536 210	6128 029
WI 6	Central West Basin	535 051	6127 398
WI 7	Near Denmark River confluence	533 716	6129 630
WI 9	North of the Elbow	537 450	6127 096

Table A.3. Site coordinates as determined by GPS.

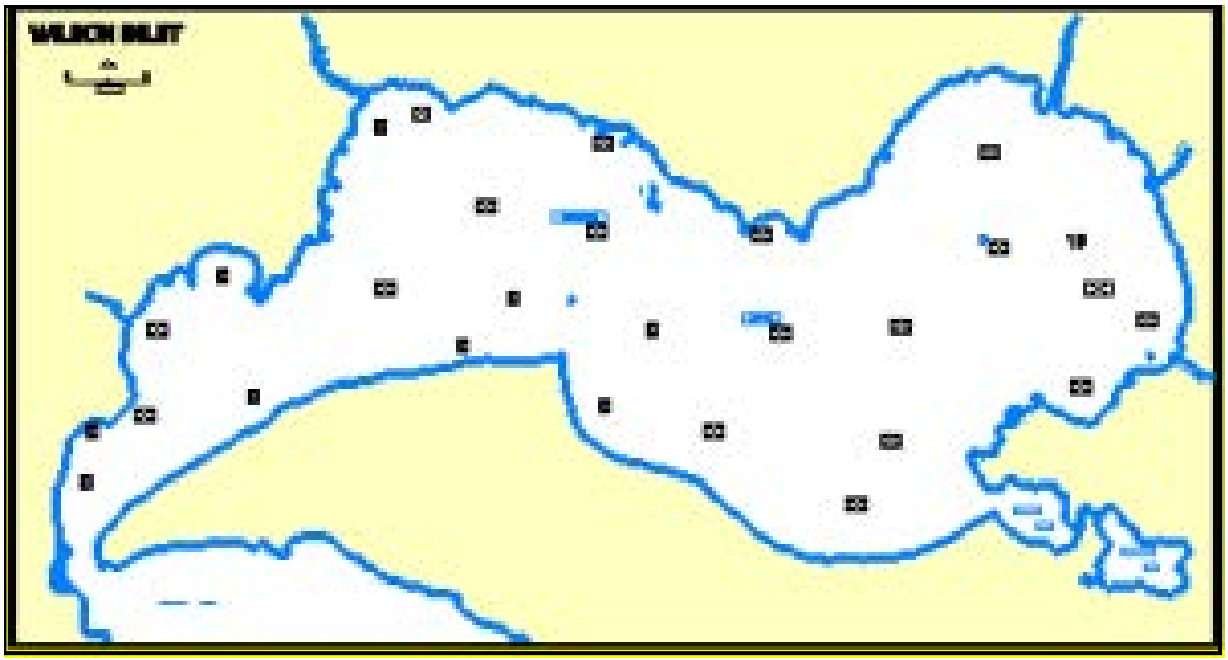

Figure A1. Map of Wilson Inlet showing location of current routine sampling sites (sites 2, 6,7,35,9,12,and 14A) as well as sites that have been monitored in the past.

Flow Data

Flow data was from the following Stations

603004 Hay River

603013 Cuppup River

603007 Sleeman River

603136 Denmark River

Daily flow volumes are in units of ML, while daily instantaneous flows are in m^3/s.

Appendix B

Load Calculations

Year	species	Basin Bottom Water Average	Equivalent Bottom Concentration	Surface Water Concentration	Total	Date	Calculation Comment
1995	NH4	0.53	0.050	0.070	0.12	12/09/95	
1995	FRP	0.025	0.002	0.002	0.004	12/09/98	one high B reading at wi 35
1995	NOx				0.12	30/08/95	inlet wide average
1995	Si				2.30	30/08/95	inlet wide average
1995	chla				0.015	27/09/95	inlet wide average
1996	NH4	0.31	0.029	0.025	0.05	2/09/96	
1996	FRP				0.009	25/10/96	inlet wide average
1996	NOx				0.27	22/08/96	inlet wide average
1996	Si				1.72	22/08/96	inlet wide average
1996	chla				0.01	9&17/10/96	inlet wide average
1997	NH4	0.17	0.016	0.040	0.06	17/09/97	
1997	FRP				0.011	17/09/97	inlet wide average
1997	NOx				0.06	17/09/97	inlet wide average
1997	Si				3.00	17/09/97	inlet wide average
1997	chla				0.01	12/10/97	inlet wide average

Appendix C

Dates of Bar Opening

Opened	Closed
22 July 1991	28 March 92
3 August 1992	
1 July 1993	18 January 1994
16 July 1994	December
19 August 1995	16 February 1996
7 August 1996	23 February 1997
18 August 1997	19 January 1998
10 August 1998	