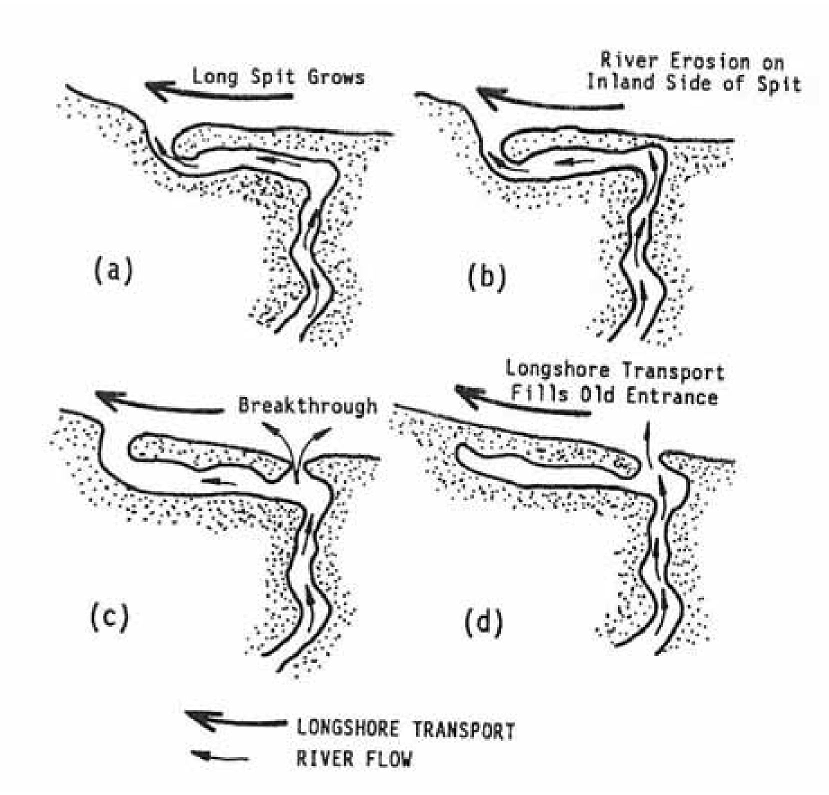

Diagram 6. Spit development and entrance breakthrough (source: NSW Government, 1990)

A major flood in 1893 broke through to the ocean north of South West Rocks. By 1901 this new entrance had been formalised by extensive channel excavations by Public Works and the closing of the original entrance, although it is not clear whether closing of the entrance referred to physical closing or closing for shipping movement. The parish maps of the early 1900's show the river to still enter the ocean south of Grassy Head, as well as at South West Rocks, yet there is no mention of the entrance remaining open in the historical description by Telfer (2005).

The parish maps show the Macleay Arm did not changed significantly between the Coode drawings of 1890 and the early 1900's, although the accuracy of both is unknown. The Arm was relatively straight and is shown further towards the east with a narrow strip of land separating it from the ocean. Near the entrance, the river curved toward the west, with a sand spit to the south of the entrance. A number of flood runners appear along the western side of the river, as well as a small island south of the entrance.

To the north of Fisherman's Island at the southern end of Shark Island the river appears to have moved westward since the early 1900's, with erosion of the outside bank and deposition along the inside bank. There has also been some additional deposition along the inside bank between 1956 and 2003. The parish maps show three islands near the entrance to Clybucca, whereas the 1956 and 2003 maps show four islands. North to south these are Muzzers Island, Snake Island, Whisky Island and Little Shark Island. The two southern islands shown in the parish maps are significantly different in size and shape from the current islands, whilst Muzzers Island shows less change in overall shape. Muzzers Island appears to have significantly decreased in size between the parish maps and 1956 and increased slightly on the western bank between 1956 and 2003.

What is now Snake Island and Whisky Island used to be a single island, long and narrow in shape except for at the southern end. Little Shark Island has significantly increased in size since the parish maps and has also extended further south between 1956 and 2003. The extensive shoaling in the area suggests that this was a depositional area for the major entrance channel dredging that continued until the 1940's. The deposition is likely to accompanied by erosion of the banks or bed to maintain the conveyance of the channel. The left bank to the south of Clybucca Creek has already eroded between 1956 and 2003.

The island to the north west of the entrance has also significantly increased in size between the early 1900's and 2003. To compensate for this increase, the outside banks of the channel have eroded in the vicinity of the island. However, further north, the channel has contracted, which is likely to be an adjustment in response to the reduced flow along the Macleay Arm after the movement of the entrance to South West Rocks.

The parish maps show South West Rocks Creek connected to the Macleay River near Ch 1. The creek is in a significantly different location compared to the location in 1956 and 2003, being further north and much wider. The parish maps do not show sufficient detail to determine the creek's shape near the ocean and it is unclear whether it was connected to the ocean at the time. Given the low topography of the surrounding swamps it is possible that the creek may form new paths during major floods, although some mapping error may also have occurred. By 1956 the creek had become separated from the Macleay River and appears to have been connected to the ocean. There does not appear to have been a significant change in planiform between 1956 and 2003, although the river location is difficult to determine from the 1956 aerial which is highly pixilated and very dark. The low relief also makes it difficult to determine the location of the top of bank in both the 1956 and 2003 aerials as the top of bank is highly dependent upon the water level in the creek at the time. Much of the creek is shoaled, with extensive shoaling just upstream of the entrance. This has been included within the river profile, however it is unknown at what water level it becomes covered.

The southern tip of Shark Island has been eroding since the early 1900's and between 1956 and 2003 a section has become separated from the main island to become a smaller island.

4.6. Coastal Change

Coastlines are continuously changing as a result of the sea, atmosphere, and coastal river processes. Waves can cause bank collapse and beach erosion, yet can also assist in beach formation through the deposition of offshore sediments. Along the NSW coastline, there is generally a net movement of sediment in a northerly direction due to dominant south-easterly wave conditions. Wind action can also play a significant role in the movement of sediment. Dominant south easterly winds in the Coffs Harbour area result in a general north westerly sand drift, although the magnitude of annual sand drift is predicted to be less than in other areas along the NSW coast (NSW Government, 1990).

Examination of the early 1900's parish maps and the 1956 and 2003 aerial photographs indicates that there has been a significant change to the Macleay coastline between the early 1900's and 2003. Figure 25 shows an outline of the approximate coastline over this period. Both the approximate location of the shoreline during low tide and the top of the beach have been shown. It should be noted that both of these are estimates as the tidal conditions at the time of the photos and drawings is unknown and the beach extent is dependent upon the amount of vegetation present.

Much of the observed change in shoreline can be attributed to construction of the training walls along the new entrance in 1897, which initially prevented the natural movement of sand northwards along the beach. This has resulted in an accumulation of sand down drift of the southern training wall and erosion of the beach to the north of the northern training wall.

In the early 1900's both Back Beach to the south of the entrance and the beach to the north were in line with one another. However, by 1956 approximately 220,000 m^3 of sand (assuming an approximate dune height of 3.5 m) had shifted from the southern end of Back Beach near Point Briner to the northern end of the beach south of the breakwall. Between the early 1900's and 1956, there was a significant loss of sand to the north of the breakwall, with the breakwall acting as a barrier to sand movement from further south. Within the accuracy of the Parish Map and the extent of the aerial photographs provided approximately 2,300,000 m^3 was lost over a distance of 3.2 km. On average, the shoreline retreated approximately 180 m.

Between 1956 and 2003 there was an additional accumulation of approximately 1,000,000 m^3 sand along Back Beach. However, the foredune became narrower due to the establishment of vegetation on the back dunes. To the north of the breakwall sand had accreted (by approximately 360,000 m^3), indicating that sand by-passing of the entrance was occurring. This sand movement across the entrance is also evident from the 2003 aerial photograph. Despite this more recent accumulation, there has been a net loss of approximately 4,500,000 m^3 to the north of the breakwall between the early 1900's and 2003.

4.7. Sediment Movement Summary

An examination of planiform changes between the early 1900's and 2003 indicate that the majority of Macleay River Estuary has not changed significantly over the last 100 years, suggesting it is relatively stable. The most significant changes have occurred in the lower estuary and along the coast, primarily the movement of the estuary mouth from Grassy Head to South West Rocks following a major flood in 1893. This has resulted in a westward shift and contraction in river width along the Macleay Arm as a result of infilling by aeolian sand from the beach dunes (and some dredged sand deposition). Along the NSW coastline, there is generally a net movement of sand in a northerly direction due to dominant south-easterly wave conditions. Localised wind conditions also suggest dominant south easterly winds, which contribute to the northerly movement of sand. The construction of breakwalls along the new entrance intercepted

FIGURE 25
COASTAL CHANGE

J:\Jobs\26017\ArcView\Maps\ReportFigures\Figure25_CoastalChange.mxd much of this northerly littoral zone sand movement, resulting in prograding of the coastline (seawards extension) to the south of the breakwalls and erosion/recession of the northern beach.

Extensive rock revetment along 66 km of river banks has restricted bank erosion and movement between 1956 and 2003. In areas that are free to adjust, the length and severity of the erosion is low when compared with similar nearby river estuaries. The erosion that is occurring appears mainly to be current initiated and then maintained by a combination of wind and boat waves. This is more likely to occur where the soils are highly erodible. The banks at Jerseyville, Kinchella, Fattorini Island and south west Kempsey are currently the most vulnerable to erosion.

Whilst rock revetment provides local bank stabilisation, it does not prevent the river from adjusting. By restricting river movement in an area of change, bed or bank erosion is likely to occur in surrounding unprotected areas. Whilst it is not possible to predict exactly where these changes will occur, or to what magnitude, it can be assumed that unprotected areas in close proximity to rocked areas will be affected. These changes are likely to be more pronounced in areas where there was significant movement prior to rock armouring.

It is not known how the river will continue to adjust to past changes. However, the most significant impacts are likely to be caused by major flooding, climate change, surrounding land practices and direct modifications to the river. Flooding can result in the formation of new channels and can cause major bank collapse. The majority of these changes are unpredictable, although exposed banks that are already vulnerable to erosion are more likely to be affected. Sea level rise combined with changed rainfall patterns have the potential to result in significant alterations to river processes, such as the tidal flux and the intensity and frequency of major flooding. Historical land practices have already resulted in extensive clearing of vegetation both on the floodplain and within the riparian corridor. This has the potential to increase the sediment load to the river as well as result in increased rates of bank erosion. Unless revegetation of the banks and the exclusion of cattle occurs, sedimentation and bank collapses are likely to continue.

5. WATER AND SEDIMENT QUALITY

Water and sediment quality has an impact on estuary health and can be affected by contaminants from both natural origins and human activities. When contaminants in the catchment are provided with a transport mechanism such as rainfall runoff, they can be washed into the estuary and become deposits on the river bed or become dissolved in the water column. Increased nutrients, faecal material and suspended solids from urban and rural runoff and sewerage system discharges are often the main contaminants in NSW estuaries. However, for the Macleay River Estuary, acid sulfate leachate resulting from the oxidation of pyrites (iron sulphide) in the soil is also an issue.

This chapter describes the existing water and sediment quality in the Macleay estuary and discusses the interactions between water and sediment quality and different estuarine processes. Due to the complexity and number of different interactions between estuarine processes, only those considered to be the most significant in terms of water and sediment quality were examined. These have been identified in consultation with Kempsey Shire Council and Southern Cross University and focus on issues which have not already been adequately addressed in previous studies.

The current study involved the collection and analysis of water and sediment samples and a review of previous studies. This chapter provides an overview of the following:

- existing water quality in the Macleay estuary and a comparison with historical data where available;
- an investigation into the impacts of septic tank effluent on water quality in the Macleay Arm;
- the derivation of a nutrient budget for the estuary;
- an investigation into the bioavailability of arsenic and antimony in sediments in the Estuary;
- a summary of acid sulfate soils; and
- a description of the process interactions.

5.1. Overview of Water Quality in the Macleay Estuary

Water quality in the Macleay Estuary is impacted by a number of different sources including diffuse runoff from the upper and lower catchment, urban runoff, and point-source discharges from wastewater treatment plants. Of particular concern are nutrients, metal and suspended sediment loads from the upper catchment, nutrient loads from urban runoff and wastewater treatment plants, and acidic and low dissolved oxygen runoff from the lower floodplain.

No spatially or temporally consistent water quality monitoring program has been undertaken in the Macleay Estuary. However, there are a number of individual water quality data sets:

- Nitrate, phosphate and dissolved oxygen data collected in 1950 (Rochford, 1952);
- Nutrient, chlorophyll-a and physicochemical data collected in 1996 (Eyre, 2000); and
- Physico-chemical data (tide level, conductivity, temperature, dissolved oxygen, pH) collected by Kempsey Shire Council from 1997 to 2005 under a number of different programs.

The Rochford (1952) and Eyre (2000) data provide a useful baseline against which change can be considered. However, the Eyre (2000) data is over 10 years old and did not include the tidal section of the Macleay River or the Macleay Arm, which are impacted by urban runoff and point source discharges. Kempsey Shire Council's water quality data mostly deals with runoff from acid sulfate soils, which is not a focus of this study as it has been dealt with under a number of other projects including the Hotspots program. A summary of the potential sources and impacts of acid sulfate runoff has been provided in Section 5.6.

Due to the lack of data in the tidal river and Macleay Arm it was decided to undertake an additional 12 months of water quality sampling from September 2006 to August 2007. This recent (2006/2007) water quality data along with the Rochford (1952) and Eyre (2000) data were compiled, synthesized and interpreted to:

(1) describe the salinity structure, hydrography and flushing characteristics of the Macleay Estuary,
(2) describe temporal and spatial variations in dissolved oxygen, pH and nutrient and chlorophyll concentrations, including the effect of flow events on water quality of the Macleay Estuary,
(3) compare water quality changes in the Macleay Estuary from 1950 to present, and
(4) compare water quality in the Macleay Estuary with ANZECC (2000) Guidelines.

5.1.1. Methodology

Water quality data from 1996 (Eyre, 2000) and 1950 (Rochford, 1952) were reviewed and an additional 12 months of water quality sampling was undertaken from September 2006 to August 2007. To allow a realistic comparison of water quality in the Macleay Estuary over time only samples collected under similar flows, collected during a similar time after a similar sized flow event, and at a similar time of year were compared (Eyre, 1997). Adopting these criteria allowed for comparison of one set of dry season and one set of wet season data. Identical methods were used for the Eyre (2000) and recent (2006/2007) data collection programs. The Eyre (2000) and Rochford (1952) methods have previously been compared and the data are comparable if small corrections are made (Eyre, 1997). Water quality data was compared with the ANZECC (2000) guidelines, which are summarised in the following table.

Table 30 Summary of ANZECC Water Quality Guidelines

Parameter	Estuary ($\mu g\ L^{-1}$)	Source
Total Nitrogen (TN)	100-750	ANZECC/ ARMCANZ, 2000
Ammonium (NH_4^+)	15	ANZECC/ ARMCANZ, 2000
Nitrate (NO_3^-)	15	ANZECC/ARMCANZ, 2000
Total Phosphorus (TP)	10-100	ANZECC/ ARMCANZ, 2000
Chlorophyll-a	2-10	ANZECC/ARMCANZ, 2000
Dissolved Oxygen	>80% saturation	ANZECC/ARMCANZ, 2000

The following provides a summary of the 2006/2007 sampling methodology and data analysis procedure adopted.

Estuarine Sampling Program

Twelve monthly sampling runs were undertaken from September 2006 to August 2007 along both the Macleay Arm and main arm of the Macleay Estuary. Sampling runs in the Macleay Arm were usually completed within about 1 hour and were timed to start at the seawater end member (saline end where the estuary joins the ocean) about 1 hour before high tide, and progress upstream. Sampling runs in the main river were usually completed within about 3 to 4 hours and were timed to start at the seawater end member at about high tide and progress upstream with the high tide crest to the freshwater end member (freshwater end). In the Macleay Arm, 5 samples were collected at geographical locations (Figure 26). In the Macleay River, samples were collected at different salinity concentrations, equivalent to intervals of approximately 2 to 3 on the Practical Salinity Scale, from seawater to freshwater along the axial salinity gradient of the estuary. Additional samples were also collected at geographical locations in the tidal river between Smithtown and the Kempsey railway bridge when saltwater did not intrude this far (Figure 26).

Surface samples (top 30 cm) were collected using an acid washed and sample rinsed bottle (being careful not to collect the surface scum) immediately filtered through 0.45 µm cellulose acetate membrane filters (Sartorius) into acid-washed and sample-rinsed polyethylene vials. At each sample location 500 to 2000 ml of water (depending of suspended sediment load) was filtered through glass fibre filters for chlorophyll analysis, and 500 ml of water was collected for suspended sediment analysis. Each filter was placed into an acid-washed 10 ml polyethylene vial. An unfiltered sample was collected in an acid-washed and sample-rinsed polyethylene vial for total nutrient analysis. All nutrient and chlorophyll samples were frozen immediately in the dark on dry ice. All analytical procedures and errors are given in Eyre (2000).

FIGURE 26

WATER AND SEDIMENT SAMPLE LOCATIONS

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Physico-chemical parameters (salinity/ conductivity, DO, Temperature, Turbidity) were measured in-situ at the point of sampling using a Hydrolab multiprobe. Salinity profiles were also undertaken at each sample location. Conductivity was calibrated against 0.005, 0.05, and 0.5 M standard potassium chloride solutions. pH was calibrated with standard buffer solutions at pH 4 and pH 7. Dissolved oxygen was calibrated against a zero oxygen solution (sodium sulfide) and an air saturated beaker of water checked with a Winkler Titration. The Hydrolab calibration was checked at the end of each sampling run and never exceeded ±2.5% of the correct standard value for any of the calibrated parameters. As such, no corrections for instrument drift have been made.

Mixing Plots

Nutrient concentrations were plotted as a function of salinity (mixing plot) to determine if there is net gain or loss during mixing. If there is a net gain (internal/ external nutrient source) within the estuary the mixing plot will show an upward curvature above the actual mixing line, as shown in Diagram 7(B). Conversely, if there is a net loss (nutrient sink) within the estuary the mixing plot will show a downward curvature below the actual mixing line (Diagram 7(C)). Nutrients passing through the estuary conservatively (i.e. no net source or sink) will plot along the actual mixing line (Diagram 7(A)). The mixing plots were used in combination to interpret the major processes occurring in the estuary.

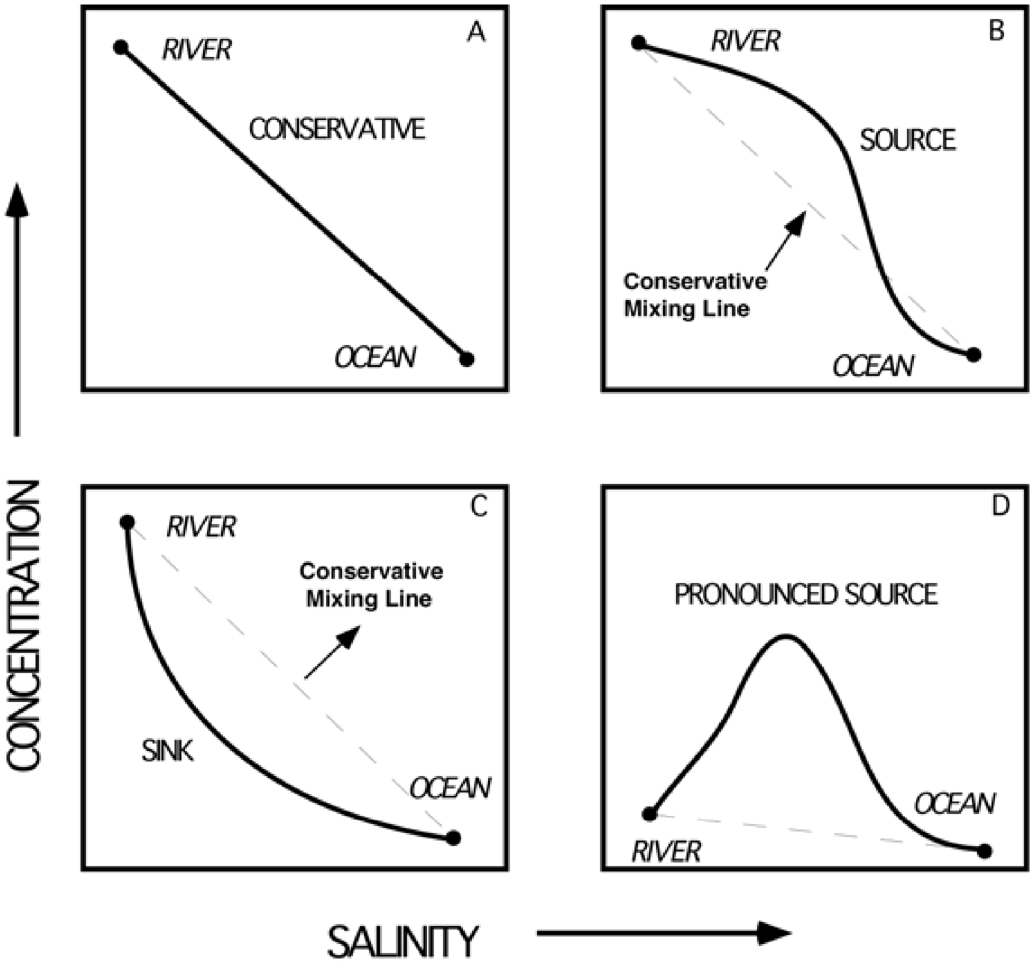

Diagram 7. Schematic representation of mixing plots showing the mixing of concentration-rich river water with concentration-poor seawater. (A) expected distribution of a substance that decreases in concentration as it moves through the estuary only due to dilution with concentration-poor seawater (i.e. it falls along the actual mixing line), (B) expected distribution of a substance if the estuary is acting as a source, (C) expected distribution of a substance if the estuary is acting as a sink, and (D) expected distribution of a substance if the estuary is acting as a pronounced source (i.e. where concentrations of the substance are higher in the estuary than the river and ocean).

Flushing Time Calculations

Flushing times were calculated using the fraction of freshwater method (Eyre 2000) using the formula:

$T = V_f/Q$

where T = Flushing time
V_f = Total volume of freshwater in a given section of the estuary
Q = Average river discharge over the period necessary to replace the freshwater in the estuary

$V_f = fd$

where f = Proportion of freshwater in a given section of the estuary
d = Volume of a given section of the estuary

$f = (S_s - S) / S_s$

where S_s = Seawater salinity
S = Salinity of a given section of the estuary

The estuary was divided into 23 boxes (section) by 24 echo sounding profiles (cross-section) at approximately 1000 to 2500 m intervals depending on the homogeneity of the estuarine section up to 40 km from the mouth (Eyre 2000). The average salinity (S) in each box (only along the salinity gradient that was inside the estuary mouth for each run) was calculated from vertical salinity profiles; lateral salinity variations were neglected. Box volumes (d) were calculated by multiplying the average of the two bounding cross-sections at high tide by the axial distance between them. The freshwater replacement time was calculated by summing daily discharges prior to the sampling date until V_f was filled. To determine the flushing time of the complete estuary, the flushing times of the individual sections were added.

5.1.2. Key Outcomes

The complete water quality data set is given in Appendix D. Only the data related to the key observations and discussion will be presented in the following sections.

Rainfall, River Discharge and Estuary Flushing Times

Yearly annual rainfall during the 2006/07 study period (1065 mm) was about 12% lower than the long-term yearly average between 1901 and 2007 (1205 mm). The lower annual rainfall during 2006/07 was due to six very dry months (Oct and Dec 2006 and Jan, April, May and July 2007), which received well below their average monthly rainfall, as shown in Diagram 8.

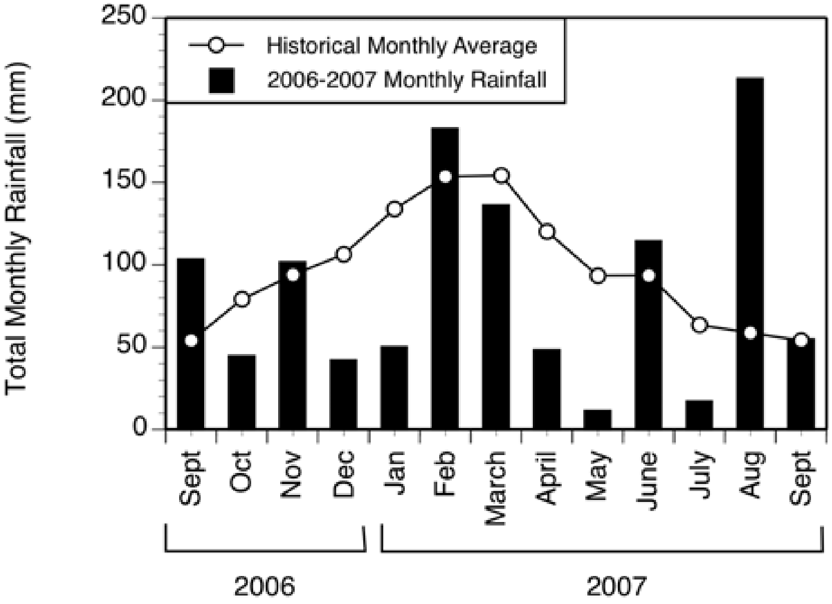

Diagram 8. Monthly rainfall during the 2006/2007 study period and the long term average monthly rainfall (1901 - 2007) for the study area.

With the exception of August 2007 (after the last sample run) which was very wet, rainfall during the remainder of the year was about average. The rain in late February, and early March 2007, resulted in a medium sized flow event in early March 2007 (28,000 ML d $^{-1}$ at Turners Flat), as shown in Diagram 9. There were also smaller flow events in September and November 2006. The sampling program covered a range of river discharges including a sampling run just after the September 2006 and March 2007 flow events and sampling during a range of dry conditions (Diagram 9).

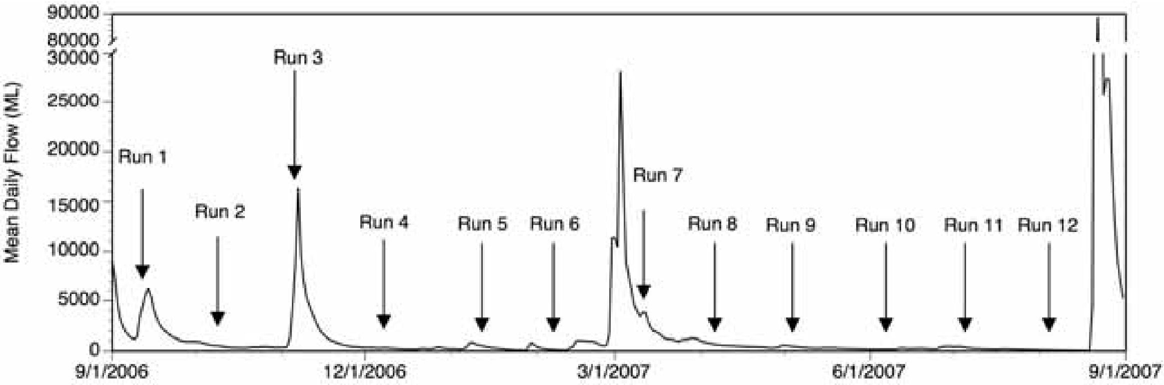

Diagram 9. Daily river discharge (Turners Flat) during the 2006/2007 study period and the timing of sampling runs.

Flushing times (

Figure 27) reflect the pattern of river discharges, ranging from about 3 days after the March 2007 flow event to about 57 days during run 11 to flush the complete estuary. A power curve was fitted to the 2006/07 data to develop an empirical relationship between the flushing time of the Macleay Estuary and the sum of the daily freshwater flow through the Turners Flat Gauging Station:

$$\text{Flushing Time (days)} = 1439 \times (\text{gauged flow (ML)} \times 1.07)^{-0.65}$$

Salinity Structure

Salinity profiles undertaken in the Macleay Estuary during 2006/07 show a full spectrum of mixing regimes from highly stratified to well mixed (Figure 28). During a number of sample runs during low flows saltwater (salinity >0.1) intruded upstream of the Kempsey railway bridge (our last sampling location) at about 40 km from the mouth. Following the March 2007 flow event the salt-freshwater interface was pushed down to 30 km from the mouth, but can be pushed out to sea during larger floods (Eyre, 2000). A power curve was fitted to the 2006/07 data to develop a empirical relationship between the distance the salt/freshwater interface intruded from the mouth of the Macleay Estuary and the sum of the daily freshwater flow through the Turners Flat Gauging Station:

$$\text{Intrusion Length (km)} = 60 \times (\text{gauged flow (ML)} \times 1.07)^{-0.08}$$

Physico-chemical

Oxygen concentrations were generally above the 80% saturation ANZECC (2000) guideline throughout most of the estuary for most runs, as shown in Figure 29. Run 7 (March 2007) immediately following the March 2007 flow event was an exception, when dissolved oxygen concentrations dropped below 70% saturation. The depressed dissolved oxygen concentrations were most likely associated with the breakdown of organic material (including NH_4 production by ammonification) mobilised by the flood waters and runoff from agricultural drains were oxygen is consumed via organic matter decomposition and oxidation of iron monosulphides (Eyre et al., 2006).

Oxygen concentrations were supersaturated in the upper estuary/tidal river during several of the runs, which reflected high rates of primary production by the dense macrophyte beds. These macrophyte beds consist of pond weeds (*Potamogeton spp*), water nymph (*Najas tenuifolia*), ribbon weed (*Vallisneria gigantea*), Clasped pondweed (*Potamogetonacea spp*), Pale Knotweed (*Polygonum lapathifolium*), Chara (*Chara spp*), dense waterweed (Egeria densa) and Elodea (Elodia canadensis) (MHL, 1997) *.* There were also supersaturated oxygen concentrations in the upper Macleay Arm during the summer runs reflecting high rates of seagrass production (runs 7 and 9) and phytoplankton production (run 6).

pH for all the sampling runs gave the expected distribution along the salinity gradient decreasing from seawater to freshwater (Figure 30). There was a slight depression of pH immediately following the March 2007 flow event. pH was elevated in the upper estuary/ tidal river during several of the runs reflecting high rates of primary production by the macrophyte beds which is

FIGURE 30

pH MIXING PLOTS SHOWING THE LOCATION OF THE GLADSTONE STP AND THE ANZECC (2000) GUIDELINES

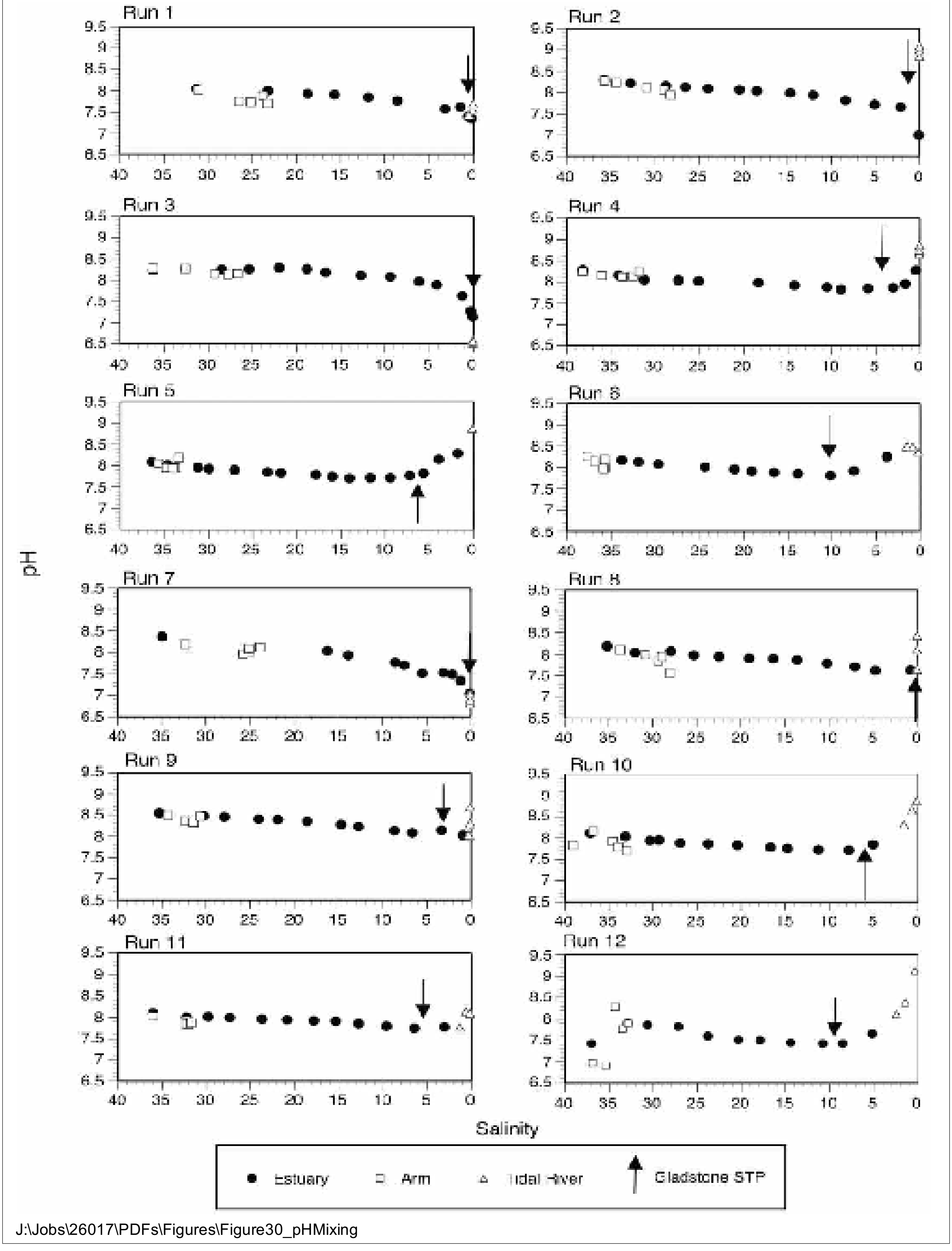

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FIGURE 28

HIGH TIDE LONGITUDINAL SALINITY DISTRIBUTION IN THE MACLEAY ESTUARY DURING EACH OF THE SAMPLING RUNS

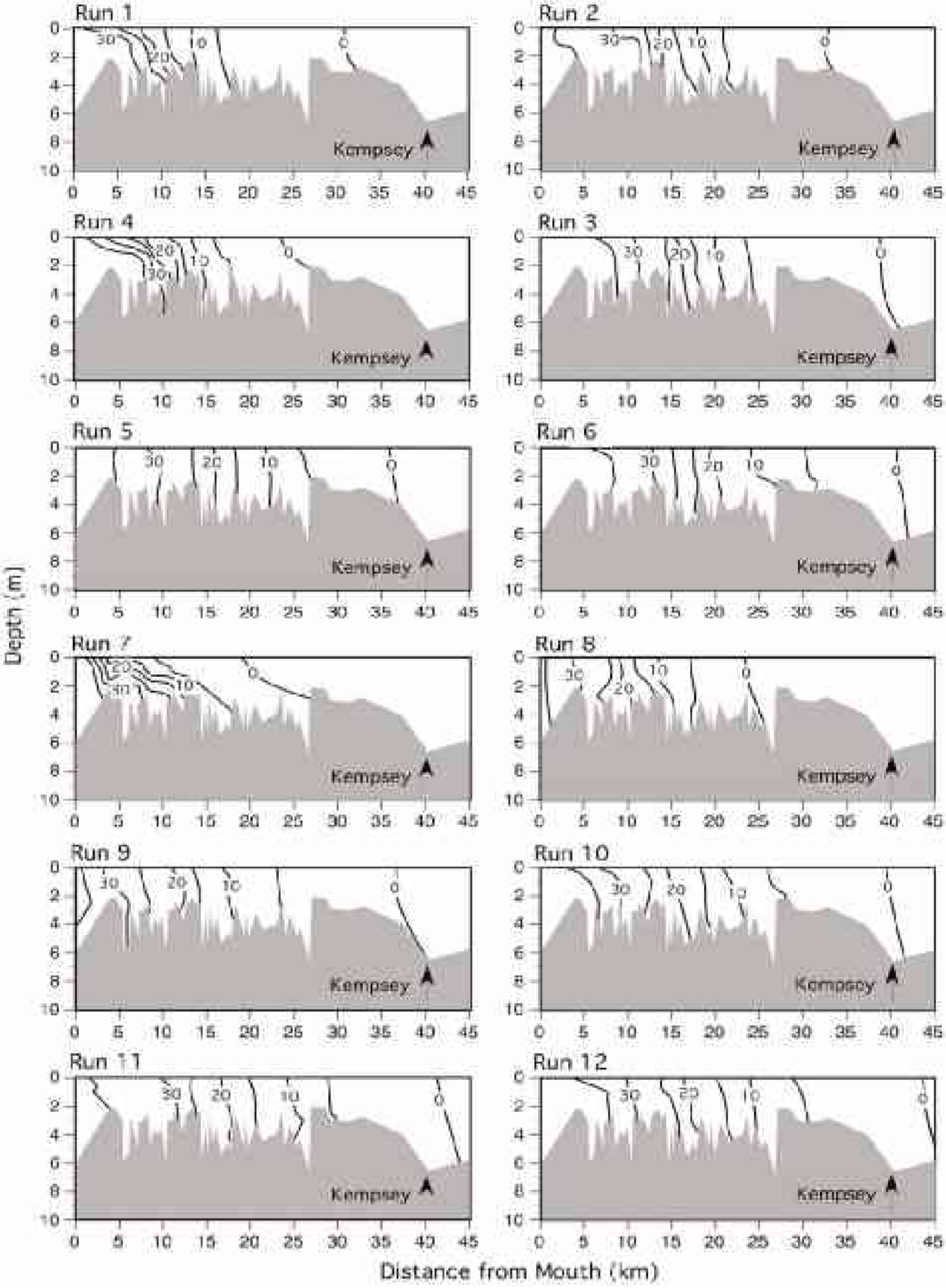

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FIGURE 29

DISSOLVED OXYGEN MIXING PLOTS SHOWING THE LOCATION OF THE GLADSTONE STP AND THE ANZECC (2000) GUIDELINES

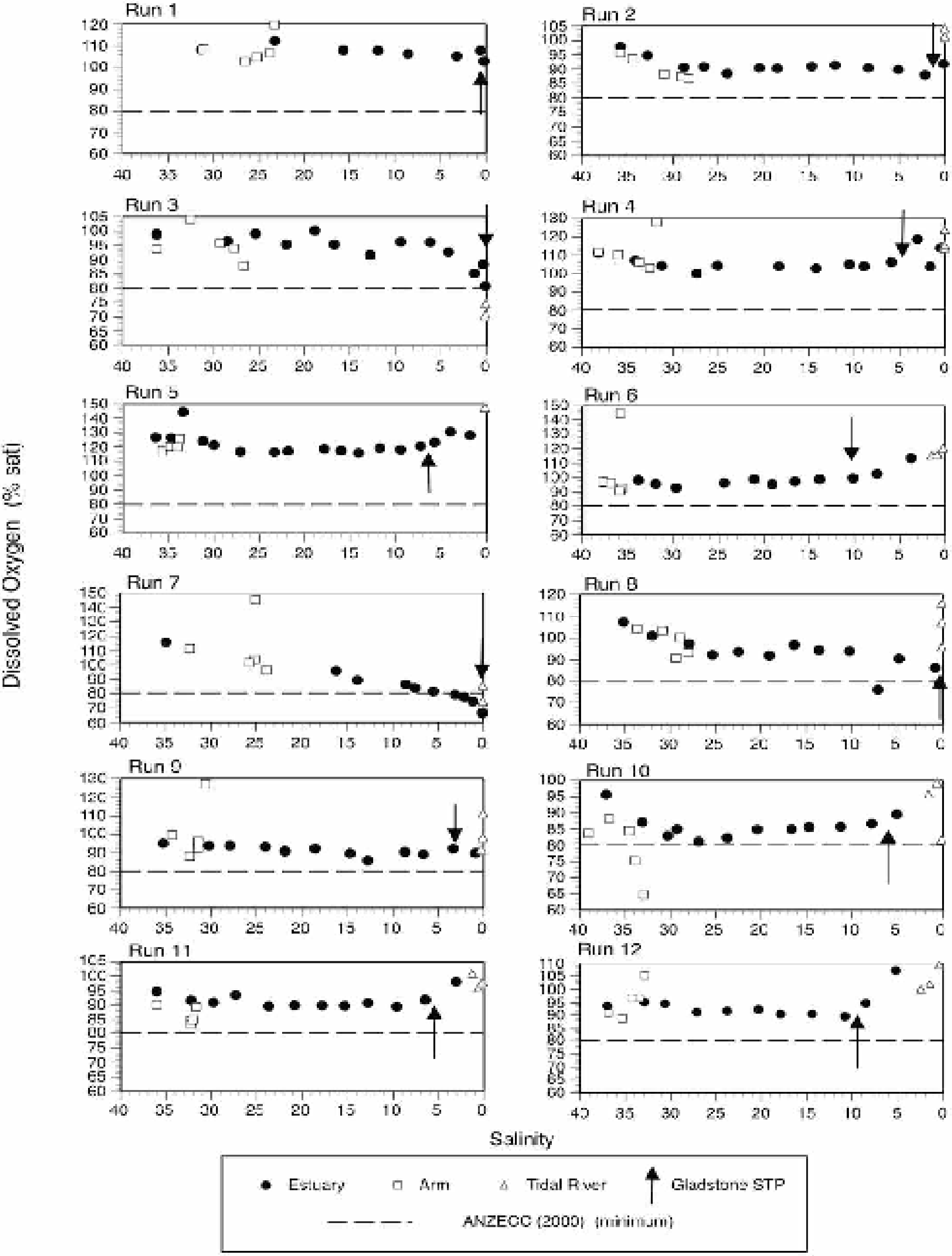

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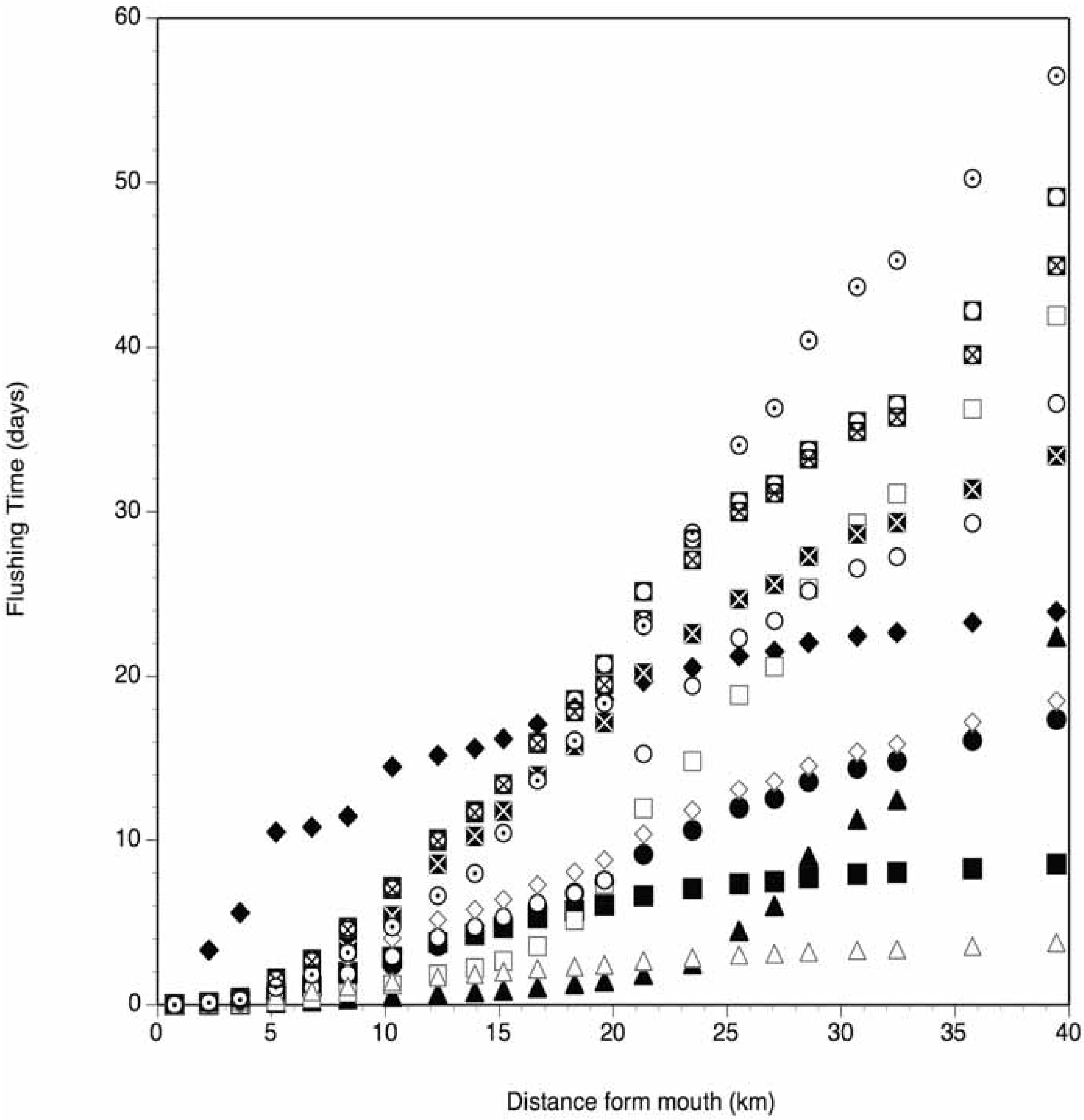
FIGURE 27

FLUSHING TIMES IN THE MACLEAY ESTUARY AS A FUNCTION OF DISTANCE FROM MOUTH DURING EACH OF THE SAMPLING RUNS



consistent with the supersaturated dissolved oxygen concentrations.

Total suspended sediment (TSS) concentrations were highest in the lower estuary and Macleay Arm due to wind resuspension, except during runs 7 and 8 following the March 2007 flow event (Figure 31). However, TSS concentrations following the flow event never exceeded TSS concentrations associated with wind resuspension. Secchi depths decreased from the seawater to the freshwater end member with the shallowest depths recorded during runs 7 and 8 following the March 2007 flow event (Figure 32).

Nutrients and Algal Biomass

Total nitrogen (TN) concentrations generally ranged between 250 and 350 µg L^{-1} throughout the estuary, which is well below the upper limit of the ANZECC (2000) guidelines (Figure 33). There was an increase in TN (mostly Dissolved Organic Nitrogen (DON)) in the lower Macleay Estuary and Macleay Arm during a number of runs. The exact source of the DON (shown in Figure 34) is unknown, but the highest concentrations were during summer (runs 4 and 5) suggesting release from seagrass beds, which can have high rates of benthic DON fluxes. There was a large increase in TN at the freshwater end member following the March 2007 flow event with concentrations up to 1200 µg L^{-1}; well above the ANZECC (2000) guidelines. Most of the TN in this diffuse runoff was delivered as dissolved organic nitrogen (DON) and nitrate (NO_3).

Nitrate (NO_3) and ammonium (NH_4) concentrations were below the ANZECC (2000) guidelines for many of the runs, except for NO_3 in the vicinity of the Gladstone wastewater discharge (Figure 35 and Figure 36). NO_3 and NH_4 concentrations also fell below the conservative mixing line for many of the runs reflecting phytoplankton uptake. During several of the runs NO_3 and NH_4 concentrations approached the detection limit suggesting phytoplankton growth in the Macleay Estuary can become nitrogen limited which is consistent with other northern NSW estuaries (Eyre, 2000).

Despite the largest wastewater nitrogen loads coming from the West and South Kempsey treatment plants that discharge into the upper estuary/tidal river, this load was not reflected in NO_3 or NH_4 concentrations suggesting these nutrients are rapidly assimilated by the macrophyte beds. The annual production by the macrophyte beds (822 t) would consume about 104 t of nitrogen (assuming a molar C:N of 10:1), which is more than the total wastewater load (18 t) from the West and South Kempsey and Frederickton sewage treatment plants. As such, wastewater nutrient loads may be helping maintain the growth of the macrophyte beds in the upper estuary/ tidal river, but these plants in turn are striping out the nutrients and making them unavailable to phytoplankton. It would be expected that algal biomass would be higher in the upper estuary/ tidal river without nutrient uptake by the macrophyte beds.

Total phosphorus (TP) concentrations generally ranged between 30 and 45 µg L^{-1} throughout the estuary, which is well below the upper limit of the ANZECC (2000) guidelines (Figure 37). During most of the dry runs phosphorus concentrations were higher at the seawater end member than the freshwater end member indicating an ocean source. Similar to TN, TP concentrations increased following the March 2007 flow event (run 7), but only just reached the upper limit of the ANZECC (2000) guidelines.

FIGURE31

TOTAL SUSPENDED SEDIMENT MIXING PLOTS SHOWING THE LOCATION OF THE GLADSTONE STP AND ANZECC (2000) GUIDELINES

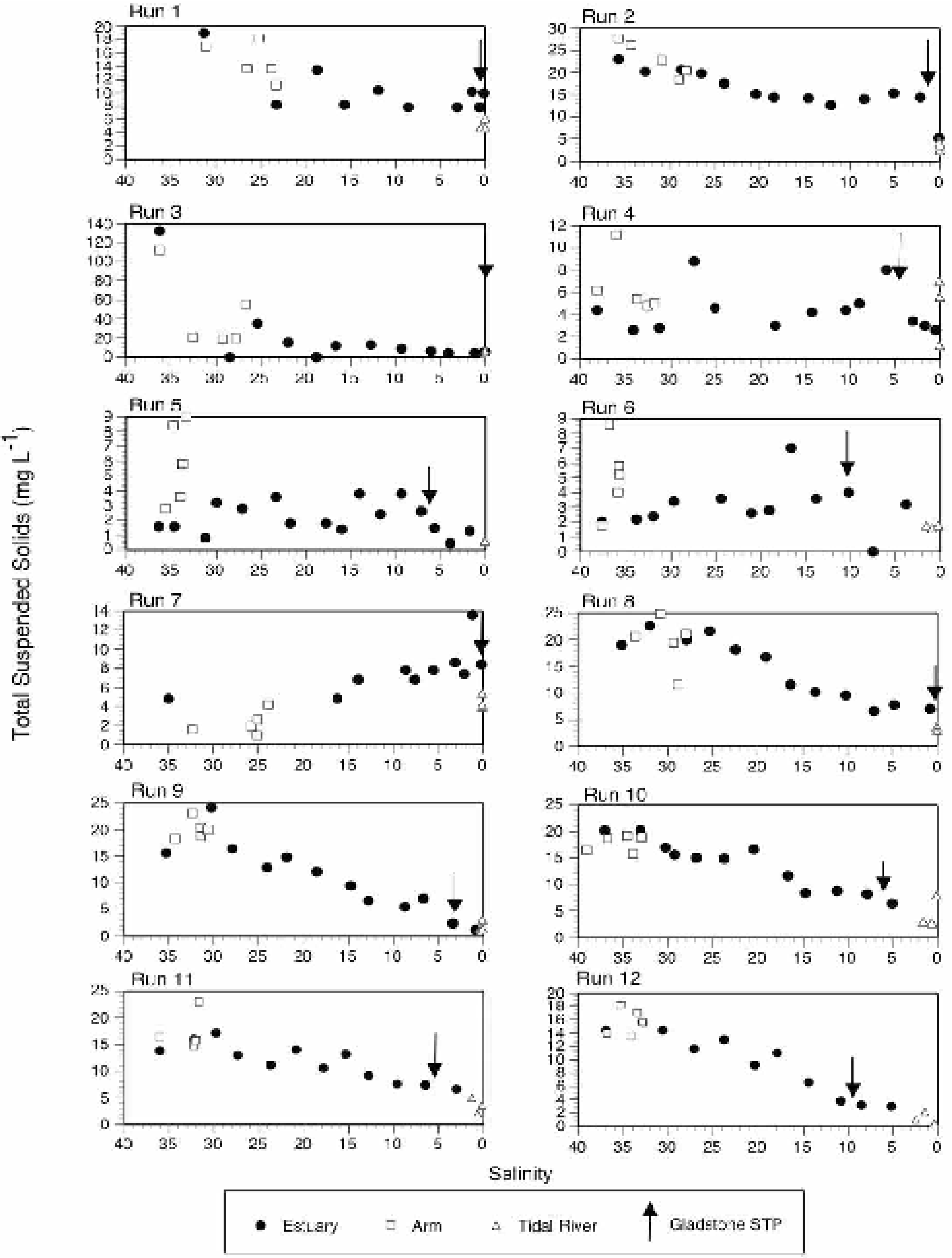


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FIGURE32

SECCHI DEPTH MIXING PLOTS SHOWING THE LOCATION OF THE GLADSTONE STP AND ANZECC (2000) GUIDELINES

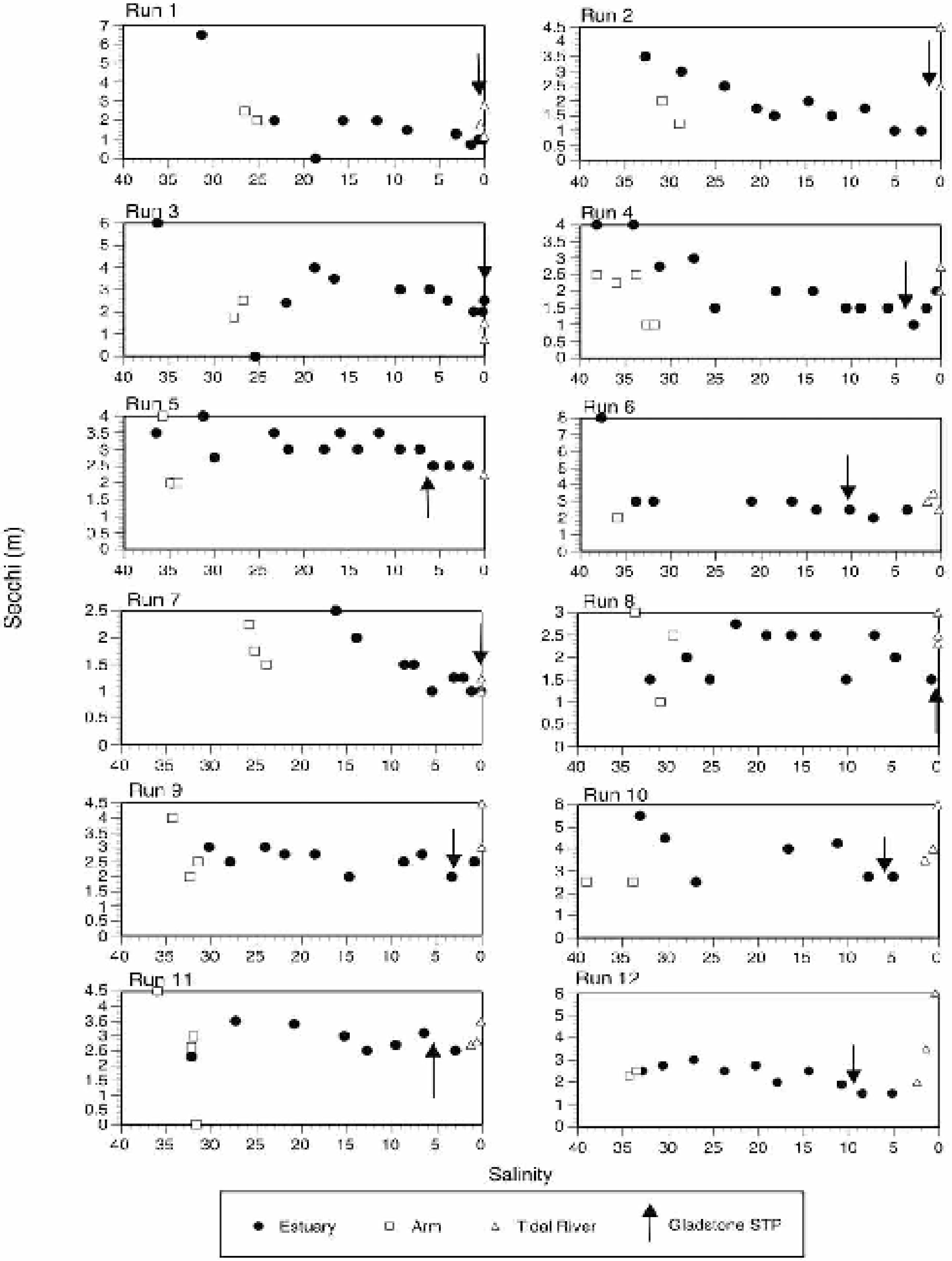


FIGURE 33

TOTAL NITROGEN (NT) MIXING PLOTS SHOWING THE LOCATION OF THE GLADSTONE STP AND ANZECC (2000) GUIDELINES

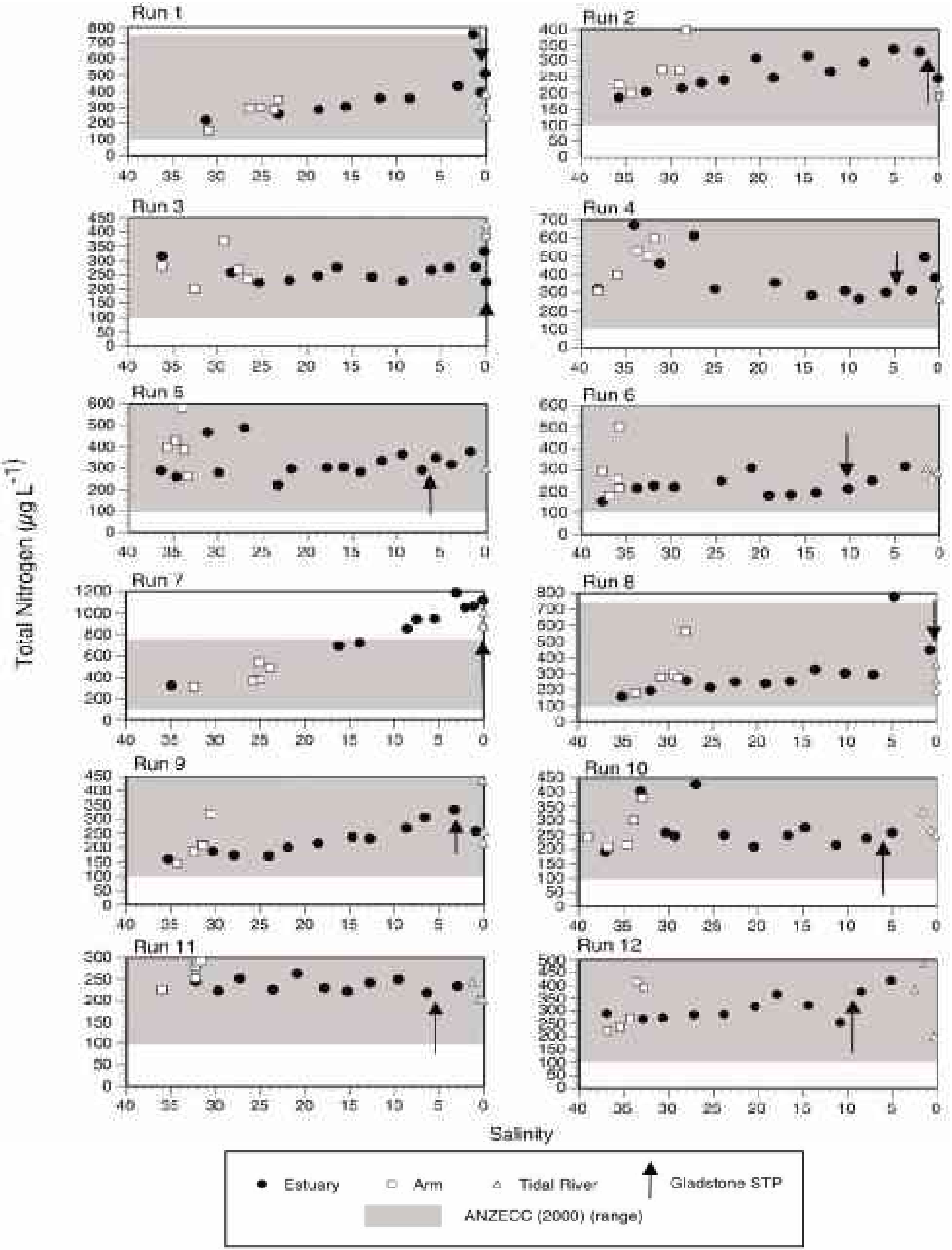
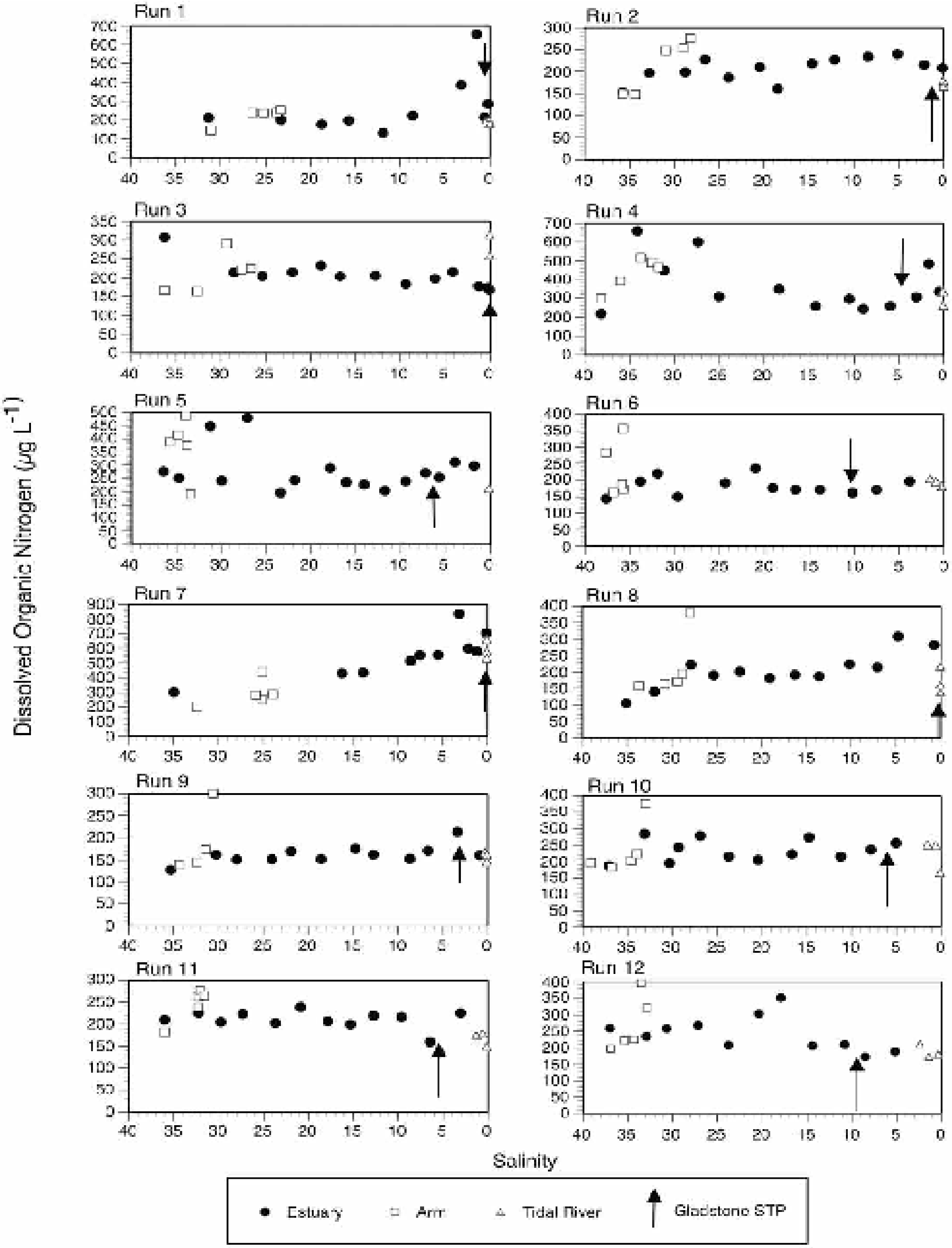


FIGURE 34

DISSOLVED ORGANIC NITROGEN (DON) MIXING PLOTS SHOWING THE LOCATION OF THE GLADSTONE STP AND ANZECC (2000) GUIDELINES



J:\Jobs\26017\PDFs\Figures\Figure34_TotalDissolvedNitrogen

FIGURE 35

NITRATE (NO_3)
MIXING PLOTS SHOWING THE LOCATION OF THE GLADSTONE STP AND ANZECC (2000) GUIDELINES

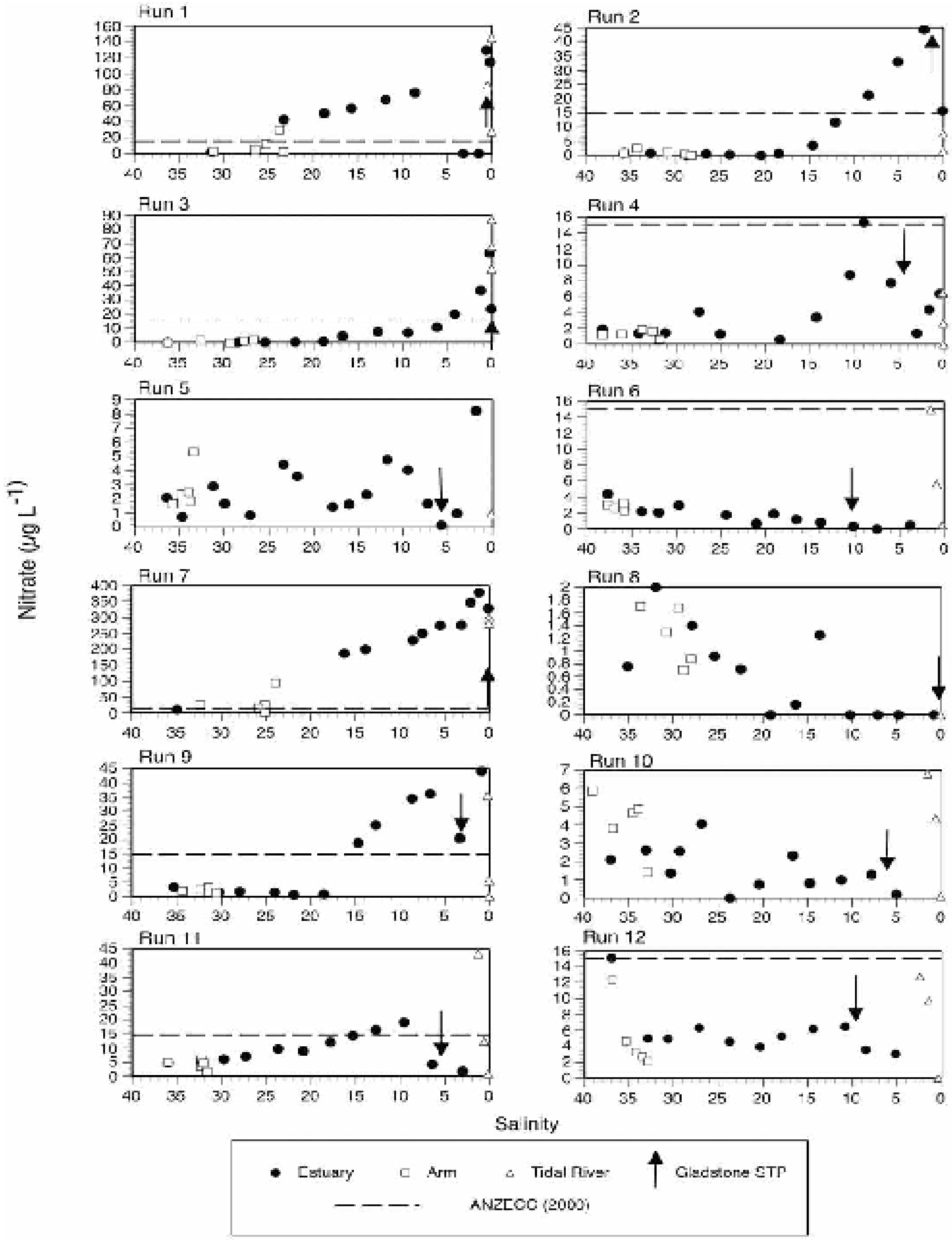


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FIGURE 36

AMMONIUM (NH_4)
MIXING PLOTS SHOWING THE LOCATION OF THE GLADSTONE STP AND ANZECC (2000) GUIDELINES

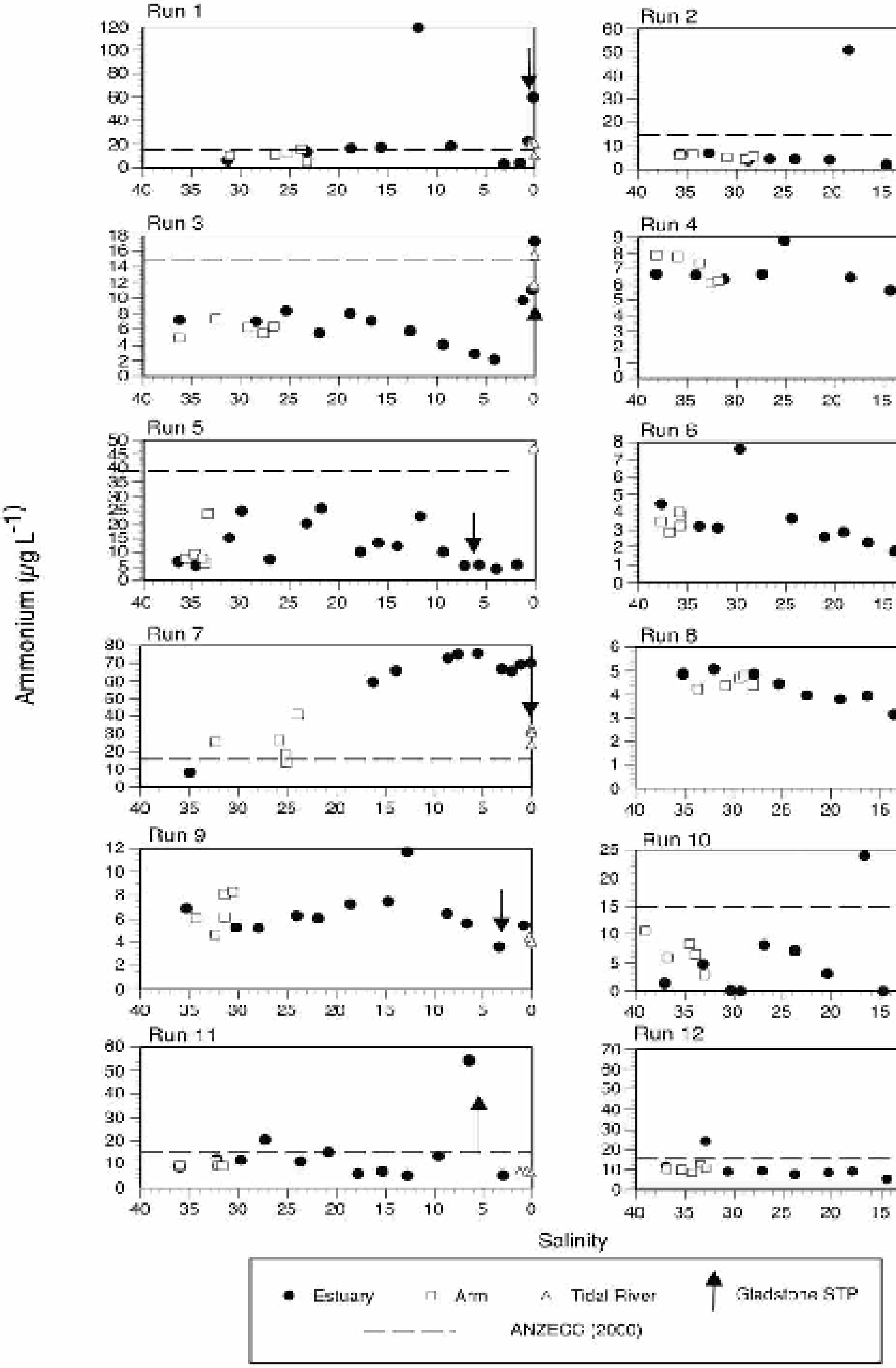

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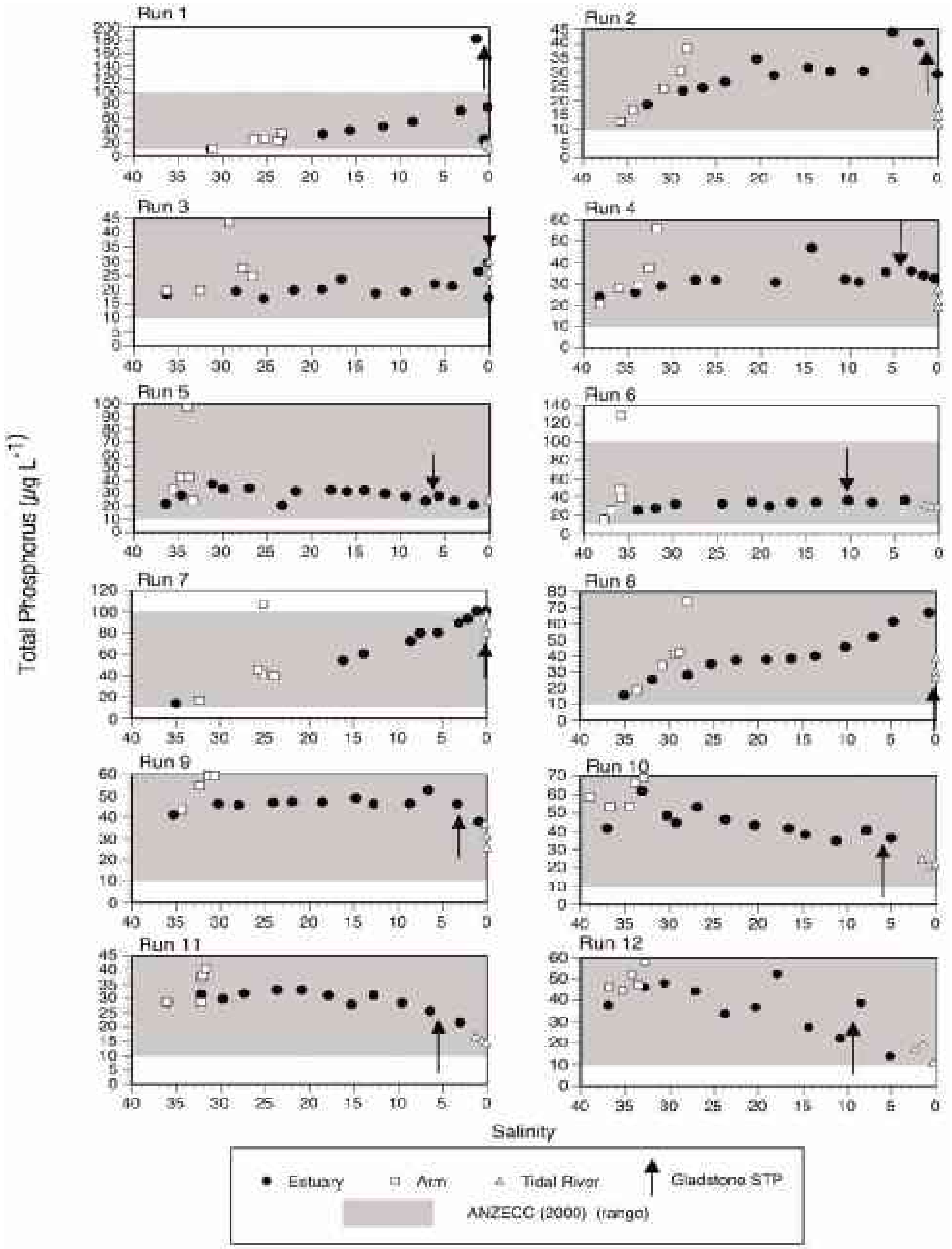
FIGURE 37

TOTAL PHOSPHORUS (TP) MIXING PLOTS SHOWING THE LOCATION OF THE GLADSTONE STP AND ANZECC (2000) GUIDELINES



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There was some uptake of dissolved inorganic phosphorus (DIP) in the upper estuary associated with phytoplankton blooms downstream of the Gladstone wastewater discharge (Figure 38), but unlike NO_3 and NH_4, concentrations never approached the detection limit, reflecting the lack of phosphorus limitation. Phosphorus concentrations were also low in the upper estuary/tidal river, despite wastewater discharges from the West and South Kempsey treatment plants, again reflecting uptake by the macrophyte beds. Phosphorus concentrations were consistently higher in the Macleay Arm than the main estuary, particularly during the summer months adjacent to Grassy Head and Stuarts Point (runs 4, 5, 6). Possible sources of this phosphorus include discharge from septic tanks associated with peak holiday loadings and/or release from the sediments due to enhanced remineralisation during summer and resuspension. A similar increase in NO_3 and NH_4 was not seen probably due to rapid assimilation by phytoplankton, which is most likely nitrogen limited. Further investigation into the possible impact of septic tanks on water quality in the Macleay Arm is discussed in the following section (5.2).

Chlorophyll-a concentrations generally ranged between 1.0 and 2.5 µg L^{-1} throughout the estuary, which is around the lower limit of the ANZECC (2000) guidelines. A number of distinct patterns emerged in the algal biomass (chlorophyll-a) data (Figure 39). There was a consistent peak in chlorophyll-a concentrations downstream of the Gladstone wastewater discharge during the dry season reflecting a point-source input of nutrients. Chlorophyll-a concentrations were low immediately following the March 2007 flow event (i.e. runs 7 and 8), despite elevated nutrient concentrations, most likely due to rapid flushing (3 days) and some light limitation. The highest chlorophyll-a concentrations occurred during run 9 in the middle reaches of the estuary most likely due to a combination of elevated nutrient concentrations following the March 2007 flow event and increasing flushing times and improving light conditions (see Secchi) as the estuary recovered post-flood.

Chlorophyll-a concentrations were consistently low in the upper estuary/tidal river most likely due to nutrient uptake by the macophyte beds resulting in nutrient limitation of algal biomass. There was a phytoplankton bloom adjacent to Grassy Head in January and February 2007 (runs 5 and 6) probably driven by an increased load of nutrients and poor flushing in the upper Macleay Arm. Possible sources of nutrients includes discharge from septic tanks, which is also consistent with the slight enrichment of ^{15}N (a nitrogen isotope) in the seagrass adjacent to Grassy Head, and/or released from the sediments during summer due to enhanced remineralization. Further investigation of the factors controlling summer phytoplankton blooms (e.g. nutrient sources) in the Macleay Arm is recommended. This program should include water column measurements of nutrients, benthic fluxes measurements and stable isotope measurements.

Historical Comparison

Dry season dissolved oxygen concentrations throughout most of the estuary showed little difference between 1949, 1996 and 2007 (Figure 40). The only exception was the upper estuary/ tidal river where high rates of production by the macrophyte beds resulted in super-saturated dissolved oxygen concentrations in 2007. However, samples were only collected in

FIGURE 39

CHLOROPHYLL-A MIXING PLOTS SHOWING THE LOCATION OF THE GLADSTONE STP AND ANZECC (2000) GUIDELINES

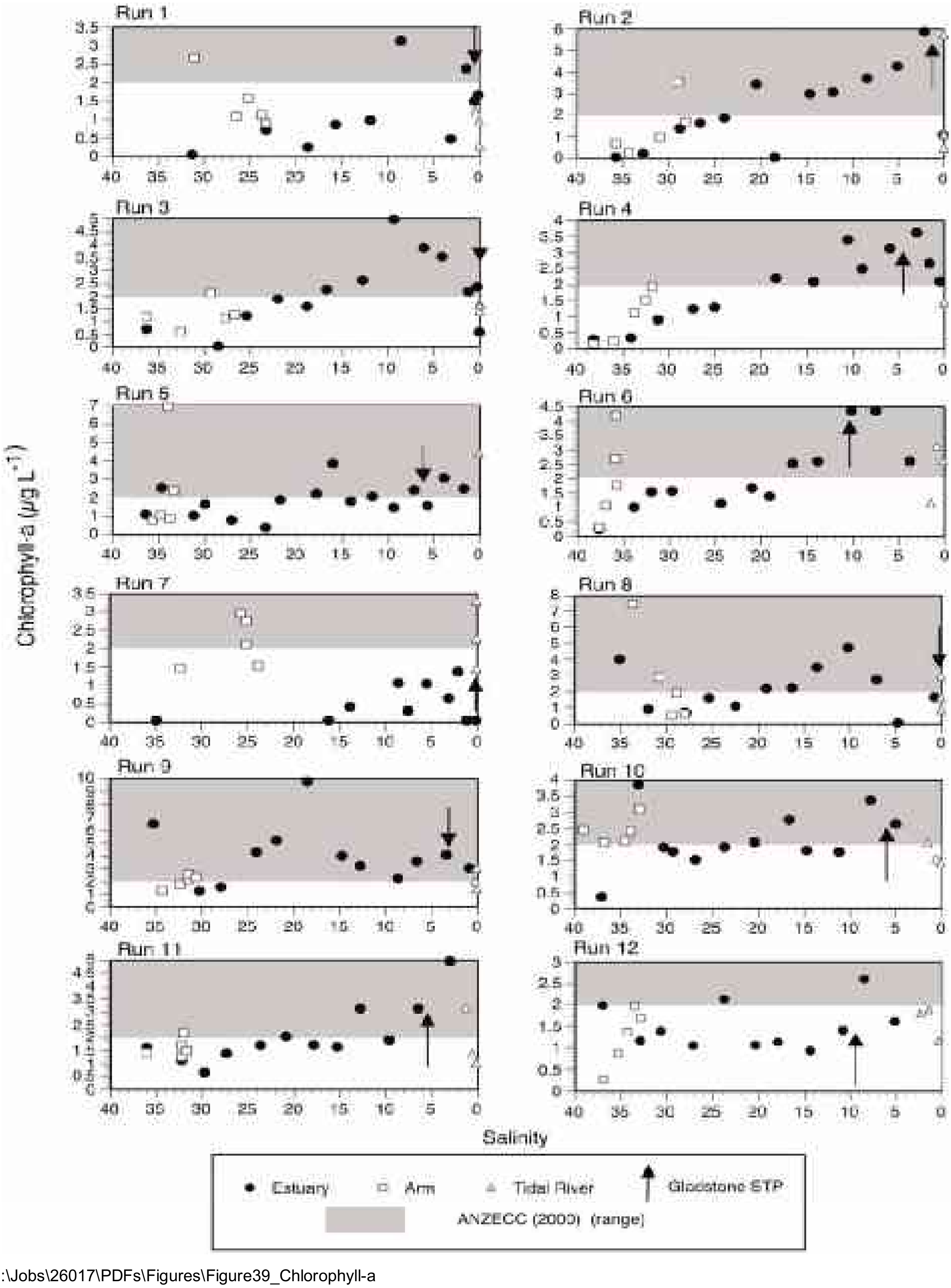

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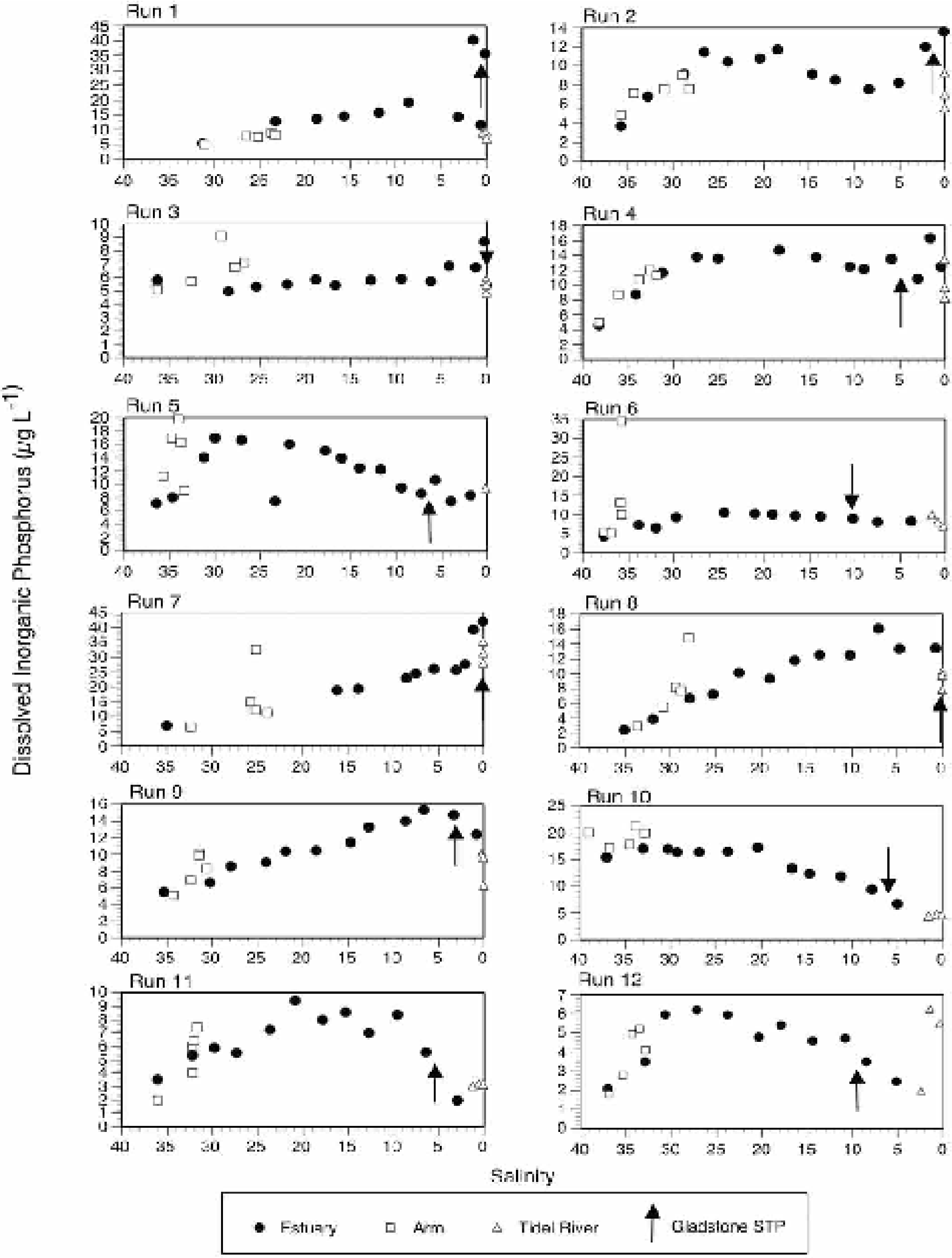
FIGURE 38

DISSOLVED INORGANIC PHOSPHORUS (DIP) MIXING PLOTS SHOWING THE LOCATION OF THE GLADSTONE STP AND ANZECC (2000) GUIDELINES



the tidal river in 1949 for comparison. There was also little difference in dissolved oxygen concentrations in the wet season between 1949 and 2007 (Figure 41). Dissolved oxygen concentrations in the wet season in 1996 were much lower than in 1949 and 2007 suggesting a different source of floodwater that had been deoxygenated via organic matter decomposition and sulphide oxidation in agricultural areas (Eyre et al., 2006). A detailed flood analysis would be required to determine the source of this de-oxygenated water. Secchi depths in both the wet and dry seasons were also similar between 1996 and 2007. TSS concentrations in the wet season were similar between 1996 and 2007, but much higher in the dry season in 2007, probably due to wind resuspension.

TN concentrations in the dry season were slightly higher in 2007 than 1996, mostly due to higher DON concentrations (Figure 40). All the inorganic forms of nitrogen in the dry season showed similar concentrations between 1996 and 2007. TN, DON and NH_4 concentrations were also similar in the wet season between 1996 and 2007, but there were some differences in the other nitrogen fractions. NO_3 concentrations at the freshwater end member in the wet season were much higher in 2007 compared to 1996 and 1946, but particulate nitrogen (PN) concentrations were much lower. The higher NO_3 concentrations in 2007 most likely reflect an increase in diffuse nitrogen loading. NO_3 concentrations in the river at Turners Flat also increased over the last 10 years (Figure 42). It is unknown why the PN concentrations were so low in 2007 as TSS concentrations were similar.

TP concentrations in the dry season were much higher in 2007 compared to 1996 entirely due to higher dissolved organic phosphorus (DOP) concentrations (Figure 40). Dry season DIP concentrations, between 1949, 1996 and 2007, and particulate phosphorus (PP) concentrations between 1996 and 2007, were identical, reflecting the role internal geochemical water/ particle interactions play in controlling phosphate concentrations (Eyre, 1997). Wet season TP concentrations were slightly higher at the freshwater end member in 2007 compared to 1996 again due to high DOP concentrations. DIP and PP concentrations in the wet season in 1996 and 2007 were similar, but much higher than in 1949 at the freshwater end member. Similar to NO_3, the increased DIP concentrations at the freshwater end member most likely reflect an increase in diffuse phosphorus loading.

Despite a possible increase in the loading of NO_3 in the wet season, chlorophyll-a concentrations were similar between 1996 and 2007 (Figure 41). Dry season chlorophyll-a concentrations were higher in 2007 compared to 1996 in the mid reaches of the estuary, despite similar nitrogen and phosphorus concentrations and light conditions.

5.2. Impacts of Septic Tank Effluent on Water Quality in the Macleay Arm

Concern has been raised that septic tank effluent discharging into sandy soils from the townships of Stuart Point and Grassy Head may be impacting on water quality in the Macleay Arm. A preliminary survey of stable isotope signatures in seagrass was used to determine if septic tank effluent may be being biologically used along the Macleay Arm. The technique uses the ratio of ^{15}N to ^{14}N in dried plant material, compared to a worldwide standard, to determine the delta ^{15}N in the plant. As sewage effluent decomposes in the septic tank, bacteria have a

FIGURE 42

COMPARISON OF NITRATE (NO_3) CONCENTRATIONS VERSUS FLOW AT TURNERS FLAT BETWEEN 1996 AND 2007

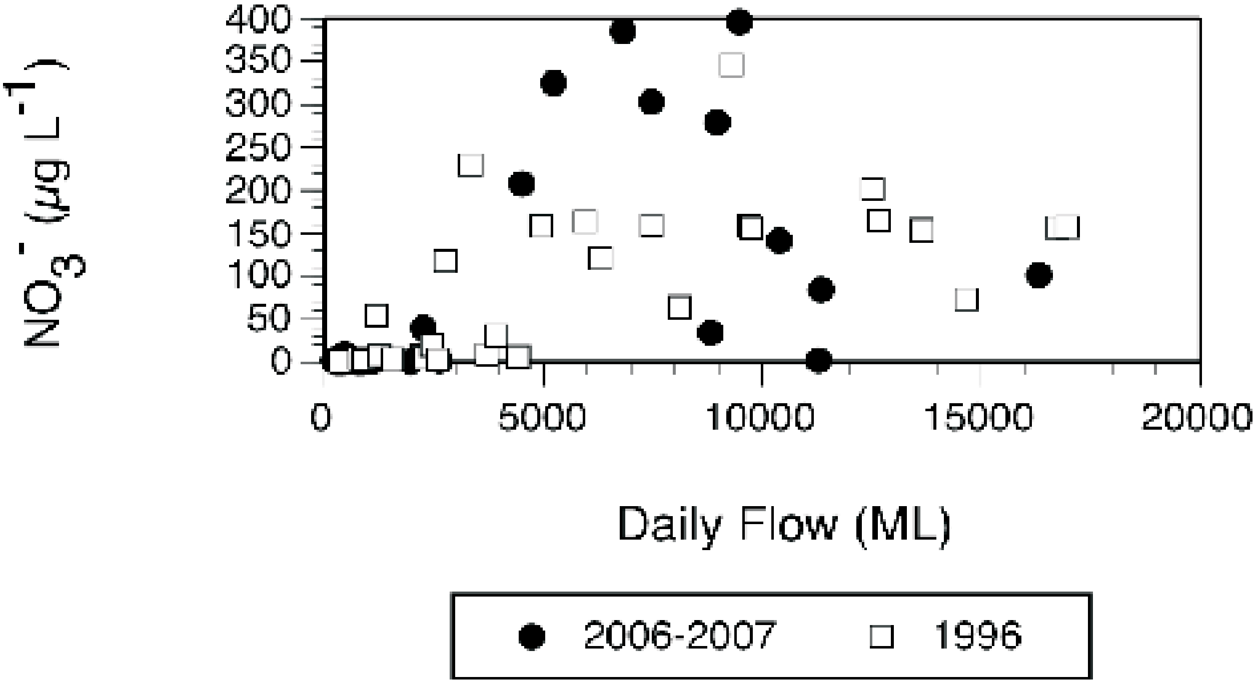


FIGURE 41

COMPARISON OF WET SEASON WATER QUALITY IN THE MACLEAY RIVER BETWEEN 1949, 1996 AND 2007

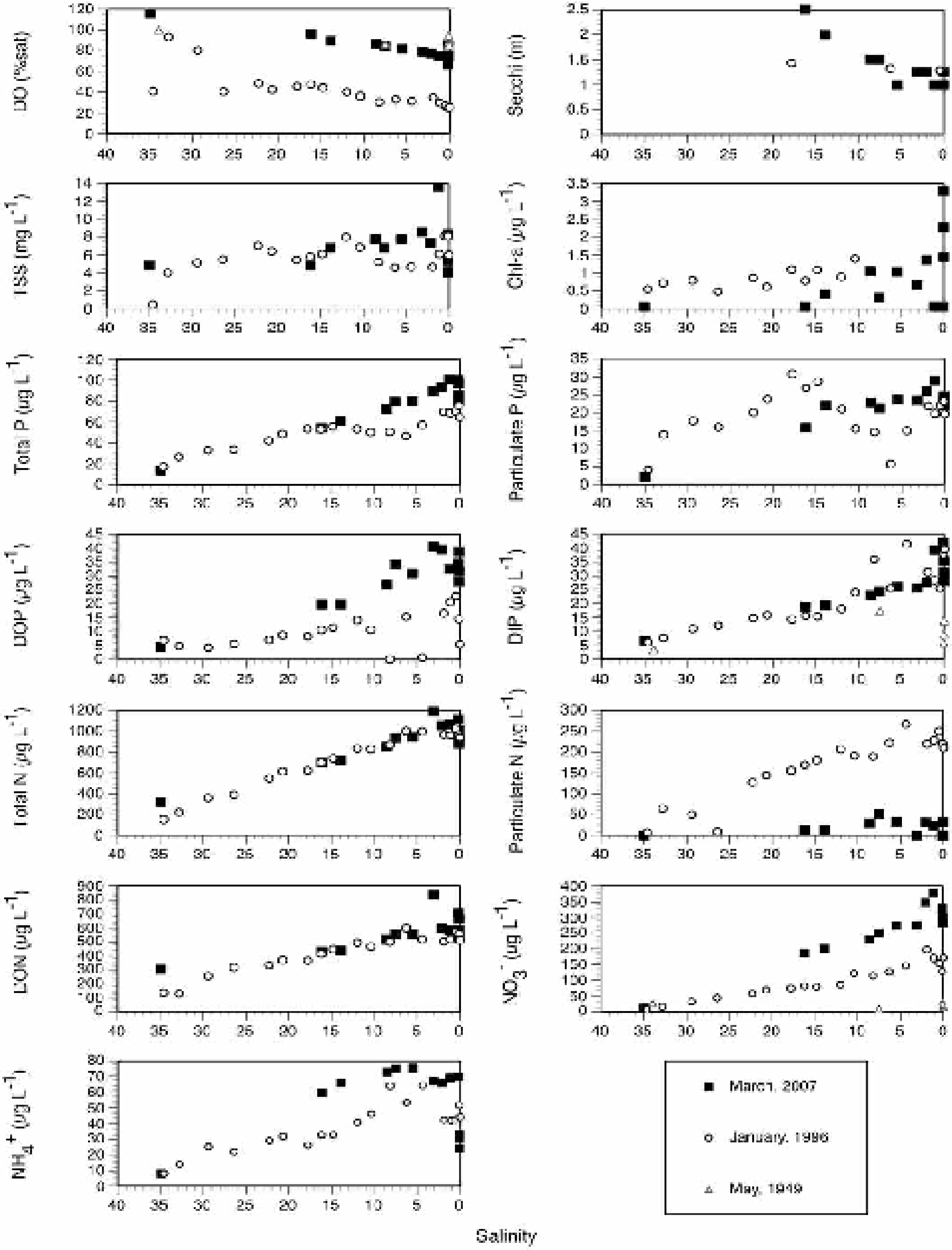
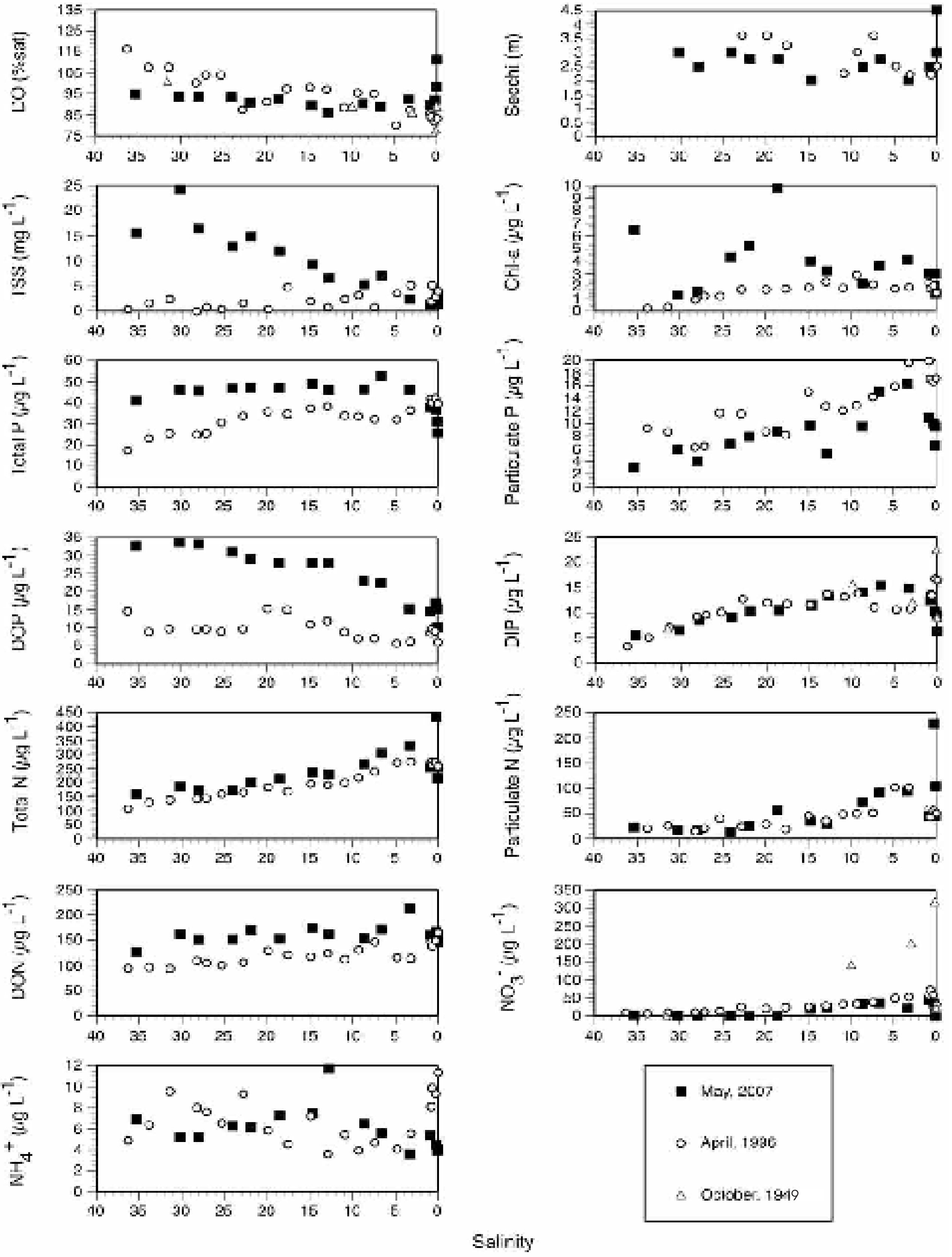


FIGURE 40

COMPARISON OF DRY SEASON WATER QUALITY IN THE MACLEAY RIVER BETWEEN 1949, 1996 AND 2007



preference towards ^{14}N over ^{15}N, making the effluent enriched in ^{15}N. When plants such as seagrass utilize this sewage nitrogen they become enriched in ^{15}N resulting in a distinct ^{15}N signature in these plants.

5.2.1. Methodology

Delta ^{15}N was measured in 6 seagrass samples collected along the Macleay Arm, as shown in Figure 26 (page 69). Sediment seagrass samples were wrapped in aluminium foil and placed in plastic bags and stored on ice until frozen at –20 ^{0}C in the laboratory. The seagrass samples were freeze-dried, ground to a powder and weighed to appropriate weights (i.e. approximately 0.20 mg for total carbon and 0.10 mg for nitrogen) and sealed in tin capsules. Capsulated samples were analysed by Continuous flow - Combustion - Isotope Ratio Mass Spectrometry (CF-C-IRMS). The CF-C-IRMS system is comprised of a Thermo FlashEA 1112 (combustion column at 1020°C, reduction column at 650°C, GC column at 45 °C, He carrier gas at a flow rate of approx. 100ml/min) interfaced through a Thermo Conflo-III to a Thermo Delta V Plus IRMS. Calibrated reference gas (CO_2 and N_2) was introduced via Conflo-III interface at the start of each run. Certified reference materials (NBS-19 and NIST 8457) were analysed at the start, middle, and end of each run as required by laboratory working standard AT-1 (Acetanilide, -1.58 +/- 0.2 delta^{15}N; -26.04 delta ^{13}C (n=20) calibrated against above certified reference materials).

5.2.2. Key Outcomes

The $\partial^{15}N$ (delta ^{15}N) signatures of seagrass in the Macleay Arm ranged from 5.7 to 7.7 (Diagram 10) which are 2 to 3 times higher than $\partial^{15}N$ signatures in seagrasses in Eastern Moreton Bay (Dennison and Abal, 1999), but similar to seagrass in the lower Richmond River Estuary away from the direct influence of sewage effluent (Eyre et al., 2007).

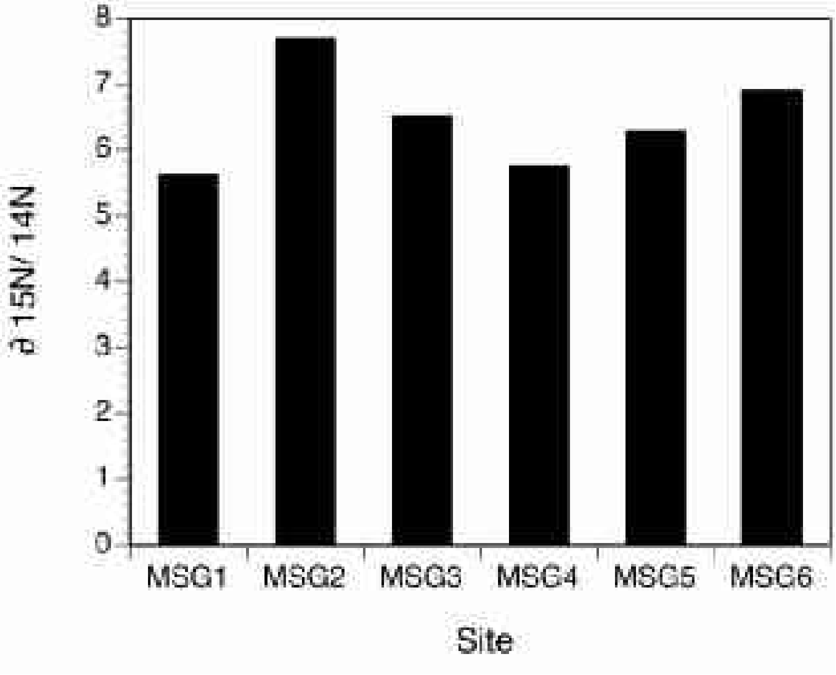


Diagram 10. ∂^{15}N values in seagrass along the Macleay Arm.

The low ∂^{15}N values in Eastern Moreton Bay away from the influence of sewage effluent compared to the Macleay Arm and the Richmond River Estuary reflects either a greater input of nitrogen via N-fixation in Moreton Bay, or the incorporation of some nitrogen from sewage effluent may be incorporated into vegetation at the sites in the Richmond and Macleay. The highest ∂^{15}N value in the seagrass was adjacent to Grassy Head, possibly indicating a greater use of sewage nitrogen at this location. The elevated ∂^{15}N value in the seagrass, combined with elevated nutrient concentrations and algal blooms (shown in Figure 39), indicates that further investigation of the input of septic tank effluent into the upper Macleay Arm is warranted.

5.3. Nutrient Budgets

Nutrient loads from a number of different sources are impacting on water quality in the Macleay Estuary (as discussed in Section 5.1). A nutrient budget is a way of quantifying the nutrient load contribution from each of the different sources. This then allows management actions to be directed towards the key problem areas. To illustrate the nutrient inputs to, and nutrient outputs from, the Macleay Estuary carbon, nitrogen and phosphorus budgets were calculated using the same framework as Eyre and McKee (2002).