Long-term variation in the water quality of the southern metropolitan coastal waters of Perth, Western Australia

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Abstract

Long-term variations in physical, chemical and biological indicators of the water quality of Cockburn Sound, Warnbro Sound and Sepia Depression were assessed by comparing data collected every two weeks over an annual cycle in 1991/92 with monthly data collected over a four year period between 1977 and 1981. In addition, trends in chlorophyll *a* and light attenuation coefficient data collected during regular 'summer monitoring programmes' (December - March) between 1977 and 1994 were examined. Inorganic nitrogen to phosphorus ratios in the water suggest that nitrogen is the primary macro-nutrient limiting plant growth in these waters, but bio-available phosphorus is also in low concentrations and may limit growth on occasions. Analytical and ecological considerations indicate that chlorophyll *a* concentrations and light attenuation coefficients are better indicators of the 'health' of oligotrophic marine waters in relation to nutrient pollution than are ambient nutrient concentrations. Based on these indicators, the water quality of Cockburn Sound was 'poor' in the late 1970s, showed a marked improvement in the early 1980s but then declined to the point where the water quality in the summer of 1993/94 was only marginally better than in the late 1970s. The initial improvement was related to reductions in total nitrogen loadings from industrial wastewater and sewage. The subsequent decline in water quality was related to a gradual increase in industrial nitrogen loads from point and diffuse sources up to 1990. The water quality of Warnbro Sound, as measured above, has remained largely unchanged since the 1970s due to the low total loadings of nitrogen to the system. The large inter-annual variation in chlorophyll *a* and light attenuation, as found in 1991/92, underline the importance of understanding natural variation when assessing the trophic status of a waterbody. Chlorophyll *a* concentrations in Sepia Depression have increased since 1984, primarily in response to the discharge of sewage from the Cape Peron outfall, and in winter, to increased nitrogen loads in the outflow of the Peel-Harvey Estuary. Nutrient levels in Cockburn Sound were significantly lower in 1991/92 compared with 1978/81 and were associated with a large reduction in nutrient loads, however the cause of the lower nutrient concentrations in Warnbro Sound and Sepia Depression is unclear. Significant relationships are presented between chlorophyll *a* concentrations and light attenuation and between nitrogen loading and phytoplankton biomass in Cockburn Sound during summer between 1977 and 1990. On the basis of these relationships current estimates of nitrogen loads to Cockburn Sound since 1990 are inaccurate. The link between nitrogen load and water clarity indicates that benthic plant communities in the oligotrophic waters of Perth are vulnerable to excessive nitrogen inputs via light limitation.

1. Introduction

Cockburn Sound and Warnbro Sound are semi-enclosed embayments bounded to the west by Sepia Depression, a long-shore trough about 5 km wide and 15-20 m deep extending from Mandurah to Rottnest Island (Figure 1). The embayments are protected from ocean swells by islands, reefs and sand banks and are poorly flushed compared to Sepia Depression (Simpson *et al.* 1993). The two embayments are characterised by central deep basins (depth > 10 m) and extensive shallow banks which, prior to the 1950s, were densely vegetated with seagrass meadows (Anon., 1979; Gordon 1986).

With the establishment of heavy industry on the eastern shore of Cockburn Sound in the early 1950s and the commissioning of a sewage outfall off Woodman Point in 1966, pollutant loads (particularly nutrients) into Cockburn Sound increased markedly. By the early 1970s concerns about the effects of waste inputs on the ecology of the Sound were raised. By this time the growth of epiphytic algae on the leaves of the seagrasses and the frequency and intensity of phytoplankton blooms had increased significantly. The combined effects of these algal blooms resulted in catastrophic losses of seagrasses, and between 1968 and 1972, approximately 80 % of the 4000 ha of seagrass meadows in Cockburn Sound were lost (Anon., 1979). Since then, the seagrass meadows in Cockburn Sound have continued to decline, albeit at a slower rate, and currently remain, in general, in a relatively poor condition (Lavery, 1994a,b).

The Cockburn Sound Environmental Study (1976-79) was initiated in response to the declining water quality and loss of seagrass in Cockburn Sound, and focussed on establishing links between the health of the aquatic environment and the natural and man-made pressures upon it. The conclusion reached was that nitrogen was the nutrient limiting algal growth in Cockburn Sound, and that a significant reduction in nitrogen load would be necessary to improve the water quality of the Sound and stop the dieback of the remaining seagrasses (Anon., 1979). This recommendation was implemented between 1982 and 1984, and involved improving the treatment of industrial wastewater and the diversion of sewage from the Woodman Point outfall which discharged into Cockburn Sound, to a pipeline discharging into approximately 22 m of water in Sepia Depression, 4 km offshore from Cape Peron.

To assess the effect of these nutrient management measures on the water quality of Cockburn Sound, a long-term monitoring programme was initiated by the Department of Conservation and Environment, now the Department of Environmental Protection, to monitor aspects of the water quality of Cockburn Sound and the nitrogen load to it during summer, every 2-3 years. This 'summer monitoring programme' (SMP) commenced in the summer of 1982/83 (Chiffings and McComb, 1983) and has continued since then (Hillman, 1986; Hillman and Bastyan, 1988; Cary *et al.* 1991; Bastyan and Paling, 1992; Bastyan *et al.* 1994). More recently, the scope of the routine SMPs have been extended to include sites in Warnbro Sound and Sepia Depression (the present study and Cary and D'Adamo, 1995).

The Cockburn Sound Environmental Study (1976-79) and the subsequent SMPs provide a long timeseries of water quality data for Cockburn Sound over a range of nutrient loading regimes. Considerably less data are available on the water quality of Warnbro Sound, although Chiffings (1979;1987) measured water quality at one site in the Sound between 1977 and 1981 as a 'control' site for contemporaneous measurements in Cockburn Sound. Industrial or domestic wastes have not been discharged directly into Warnbro Sound and, although nitrogen loads to Warnbro Sound from diffuse sources such as groundwater are estimated to have more than doubled in the past 14 years, the 1991 loads were less than 3 % of the nitrogen loading to Cockburn Sound in the same year (Martinick *et al.* 1993). Thus Warnbro Sound can be considered to be largely 'unaffected' by point source nutrient discharges and provides a useful

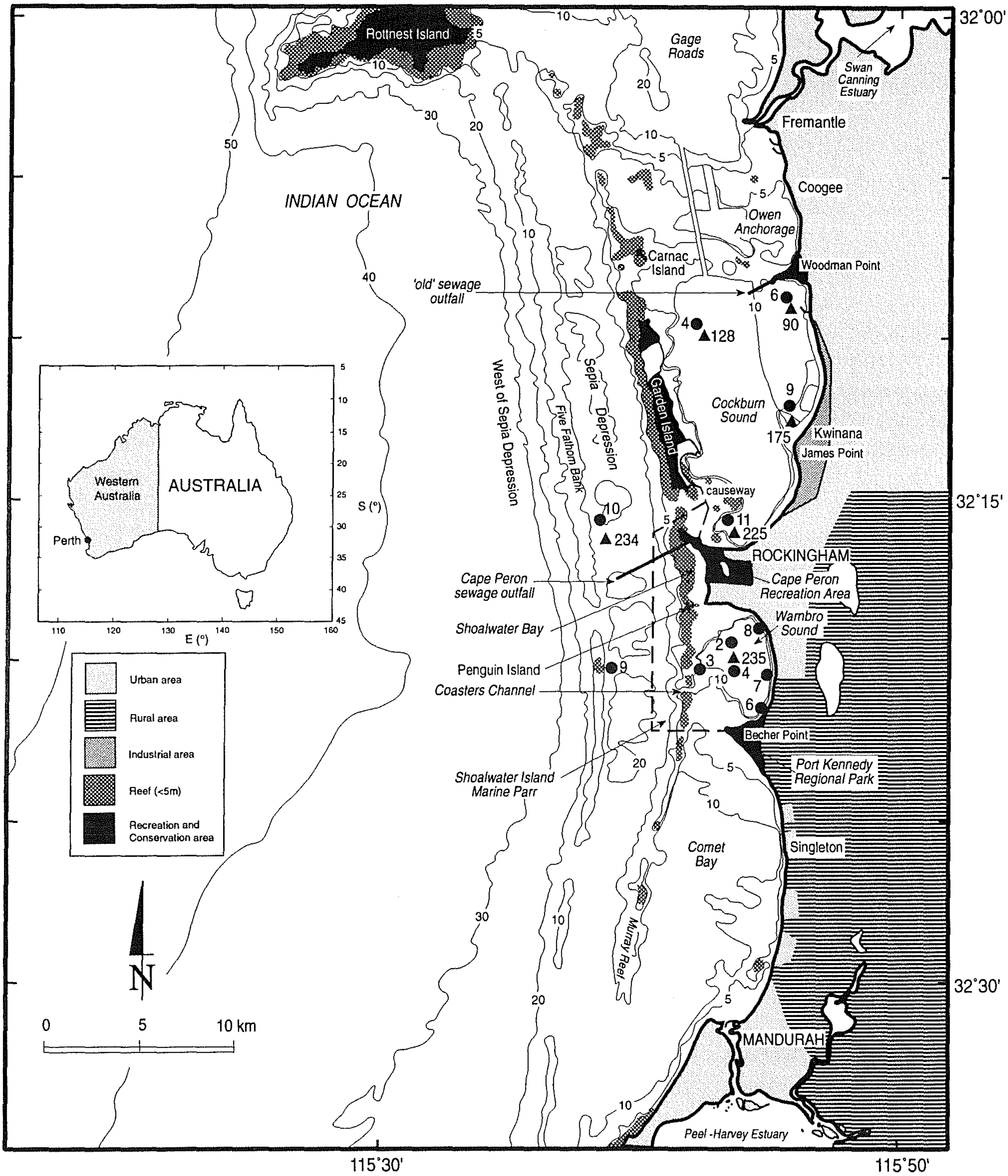

Figure 1. Location map of the southern metropolitan coastal waters study area, showing the sampling sites in Cockburn Sound in 1991/92 (sites 4, 6, 9, 11) and the 1977/81 (sites 128, 90, 225, 175), Warnbro Sound in 1991/92 (sites 2, 3, 4, 6, 7, 8) and 1977/81 (site 235) and Sepia Depression in 1991/92 (site 9) and 1977/81 (site 234). Sites 9 and 10 in Sepia Depression were sampled in 1993/94.

benchmark against which to assess the water quality of Cockburn Sound. The sewage discharged to Sepia Depression from the Cape Peron outfall is considered to be the major nitrogen source to these waters and this load has been steadily increasing since the outfall was commissioned in 1984 (Martinick *et al.* 1993). The effect of these sewage discharges on the water quality of Sepia Depression can be assessed from pre- and post-sewage monitoring programmes and the recent inclusion of this area in the SMPs. Chiffings (1979; 1987) measured the water quality at one site in Sepia Depression between 1977 and 1981, which provides some long-term perspective on the water quality of Sepia Depression.

Urban development to the south of Perth has increased rapidly during the last 10 years and this growth is expected to continue over the coming decades (DPUD, 1993) making the adjacent coastal waters increasingly important as a recreational resource, and this is reflected in the recent declarations of the Shoalwater Islands Marine Park, the Port Kennedy and Woodman Point Regional Parks and the Cape Peron Recreation Area (Figure 1). Cockburn Sound continues to receive industrial wastewater discharges (Martinick *et al.* 1993) but, as with Warnbro Sound, it also supports commercial fishing and an expanding mussel farming industry. Summer monitoring programmes in Cockburn Sound have shown that after the initial improvement in water quality that accompanied nutrient reductions in the early 1980s there was a progressive deterioration in the water quality of the Sound and this was related to gradual increases in nitrogen loadings (Cary *et al.* 1991). In addition, the remaining seagrass meadows in Cockburn Sound continued to decline, albeit at a slower rate, and most currently remain in a relatively poor condition based on 'epiphyte' and 'seagrass cover' indices (Lavery, 1994a).

The Southern Metropolitan Coastal Waters Study (SMCWS) was initiated in 1991 in response to the increasing population growth and changing patterns of usage in the area and the gradual deterioration in water quality detected through long-term monitoring of Cockburn Sound. The primary objective of the SMCWS is to understand the environmental consequences of the cumulative impacts of waste discharges to these waters, and this information will be used to develop an integrated management strategy for Cockburn Sound, Warnbro Sound and Sepia Depression consistent with maintaining the long-term health of the marine environment of the southern metropolitan coastal waters of Perth (Simpson *et al.* 1993). This present study is a component of the SMCWS and was conducted to assess the degree of long-term variation that has occurred in the water quality of Cockburn Sound, Warnbro Sound and Sepia Depression, by comparing water quality in recent years with data collected since the late 1970s and to use this information to better understand the relationships between nutrient loading and water quality in these systems.

2. Methods

2.1 Sampling regime

Water quality was sampled at six sites in Warnbro Sound (sites 2,3,4,6,7 and 8), four sites in Cockburn Sound (sites 4,6,9 and 11), and one site in Sepia Depression (site 9) (Figure 1).

In Warnbro Sound, phytoplankton biomass (as chlorophyll *a* concentration), total Kjeldahl-nitrogen (TKN) and total-phosphorus (TP) concentrations, vertical profiles of temperature, salinity and light and Secchi depth were measured weekly at all sites between January and March 1991 and approximately every 14 days between April 1991 and February 1992. In Cockburn Sound, phytoplankton biomass, temperature and light profiles and Secchi depth were measured weekly between January and March 1991 by Murdoch University (Bastyan and Paling, 1992) and used in the present study. Phytoplankton, TKN and TP concentrations, profiles of temperature, salinity and light, and Secchi depth were measured at approximately 14 day intervals between April 1991 and February 1992. At the Sepia Depression site, phytoplankton, TKN and TP concentrations, profiles of temperature, salinity and light and Secchi depth were measured weekly in March 1991 and at approximately 14 day intervals between April 1991 and February 1992. Similarly nitrate/nitrite-nitrogen (NO_3+NO_2-N), ammonium-nitrogen (NH_4-N) and orthophosphate-phosphorus (PO_4-P) were measured at approximately 14 day intervals from July 1991 to February 1992. Sampling methodologies for the 1993/94 SMP were comparable to those used in the 1991/92 SMP (Cary and D'Adamo, 1995).

2.2 Sampling procedure

Surface, middle and bottom water samples were collected by Niskin bottle (General Oceanics) and then bulked and sub-samples were taken for TKN and TP determinations. Two to six litres of the bulked water sample were filtered through a 1.2 μm G/FC Millipore filter paper at a maximum negative pressure of 75 kPa, and the filter paper retained for chlorophyll *a* analysis. Samples of the filtrate were taken for NO_3+NO_2-N, NH_4-N and PO_4-P determinations. In the field, all samples were kept in darkness on ice and were stored frozen in the laboratory until analyses were conducted. Chlorophyll *a* and nutrient analyses were conducted by the Nutrient Analysis Laboratory, School of Biological and Environmental Sciences at Murdoch University as briefly outlined below. Chlorophyll *a* concentrations (±0.06 μg l^{-1}) were determined spectrophotometrically (Jeffrey and Humphrey, 1975), PO_4-P (±4 μg l^{-1}) was analysed by the single solution method (Major *et al.* 1972), NO_3+NO_2-N (±2 μg l^{-1}) after copper-cadmium reduction with a Technicon Autoanalyser 11 (Technicon Industrial Systems, Tarry Town, New York), NH_4-N (±5 μg l^{-1}) by the phenol-prusside method (Dal Pont *et al.* 1974), TKN (±200 μg l^{-1}) after a sulphuric acid digest and analysing for NH_4-N (Anon., 1985) and TP (±10 μg l^{-1}) after a perchloric digest and analysing for PO_4-P (Anon., 1985).

Seawater temperature (±0.05 °C) and salinity (±0.05 pss) were measured at 1 m intervals through the water column using a salinity-temperature meter (Yeo-Kal Model 602). The instrument was calibrated against water of known salinity (Copenhagen Water) and against a high precision mercury thermometer. Before use the probe was soaked in 0.1 M HCl for 10 minutes to clean the platinum electrode. To account for instrument 'drift' during each cruise, water samples whose salinity had been measured in the field, were collected at the beginning and end of each day and their salinity accurately measured with an inductive salinometer in the laboratory. If 'drift' had occurred field data were adjusted accordingly. Photosynthetic Available Radiation (PAR; 400-700 nm waveband; ±5 %) was measured at 1 m intervals through the water column using an integrating quantum sensor (Li-Cor 192S) and an underwater quantum meter (Li-Cor 188B). The light attenuation coefficient was calculated as the slope of the line of best fit through the plot of $\log_{10}$ PAR versus depth and expressed in units of m^{-1}.

2.3 Data analysis

The extent of long-term changes in water quality that have occurred since 1977 were determined by analysing available data in two ways: firstly by a comparison of composite annual cycles for 1977/81 and 1991/92, and secondly by examining long-term trends in selected water quality parameters in summer between 1977 and 1994. These two approaches are outlined below.

Composite annual cycles

Data collected on 34 monthly cruises between July 1977 and June 1981 (Chiffings, 1987) and on 30 cruises over 14 months between January 1991 and March 1992 (the present study), were used to construct a composite annual cycle of monthly means of a range of water quality parameters for each of these two periods. For Cockburn Sound, four spatially similar sites to those sampled in the present study were used to construct the 1977/81 composite cycle (Figure 1). Although only one site was sampled in Warnbro Sound during 1977/81, the mean of the six Warnbro Sound water quality sites sampled in the present study were used to construct the 1991/92 annual cycle. Data were available from only one site in Sepia Depression during both periods, however the site sampled in the present study (site 9) was located approximately six kilometres south of site 235 used in 1977/81 (Figure 1) and, from comparisons with data collected between January and May from nine sites situated between 2 and 6 kilometres north of site 9 (Halpern, Glick and Maunsell, 1992), data from site 9 are considered broadly representative of the water quality of Sepia Depression. The composite annual cycles of monthly means were compared using the non-parametric Wilcoxon rank test and probabilities of less than or equal to 0.05 taken to be significant.

Long-term trends

The summer monitoring programme (SMP) data (December to March) for Cockburn Sound, Warnbro Sound and Sepia Depression were examined for long-term trends in chlorophyll *a* concentration and light attenuation. The longest continuous timeseries of water quality data in the region is from the SMP for Cockburn Sound (Chiffings and McComb, 1983; Hillman, 1986; Chiffings, 1987; Hillman and Bastyan, 1988, Cary *et al.* 1991, Bastyan and Paling, 1992; Bastyan *et al.* 1994; Cary and D'Adamo, 1995 and the present study) although, as outlined below, there has been some variation in the sampling regime used in this programme. Prior to the summer of 1991/92, water quality data for Cockburn Sound were presented as the mean of eight sites, but subsequently (except for 1992/93) the SMP was rationalised and only four sites were routinely monitored. These four sites were selected to represent the water quality of the Sound as a whole. In addition, the 1991/92 summer data for Cockburn Sound is the average of the eight weekly-cruises at eight sites (Bastyan and Paling, 1992) and six fortnightly-cruises at four sites (the present study). For Warnbro Sound the SMP data for the summers of 1977/78, 1979/80 and 1980/81 are from site 235 (Chiffings, 1987); for 1990/91 and 1991/92 from sites 2,3,4,6,7,8 (the present study); 1992/93 from sites 2 and 7 (Helleren and John, 1995) and 1993/94 from sites 2,3,7 (Cary and D'Adamo, 1995). For Sepia Depression the SMP data for the summers of 1977/78, 1979/80 and 1980/81 were collected from site 234 (Chiffings, 1987); for 1991/92 and 1993/94 from site 9 (the present study; Cary and D'Adamo, 1995).

All light attenuation coefficients presented here were calculated for the full depth of the water column. When light attenuation coefficients were only available for the top 6 m of the water column, a factor of 0.9 was applied to convert these to a full depth light attenuation coefficient.

Cape Peron sewage outfall

To identify any changes in the water quality of Sepia Depression that may have occurred since the Cape Peron sewage outfall commenced discharge in 1984, chlorophyll *a* data collected pre- and post-discharge (Table 1) were used to construct a composite annual cycle of monthly means for each of these two periods. Data for the pre- and post-discharge programmes are presented as the means of surface, middle and bottom samples from all sites, except the last study in 1992, which is the mean of surface and bottom samples. The composite annual cycles were compared using the non-parametric Wilcoxon rank test and probabilities of less than or equal to 0.05 were taken to be significant.

In addition, chlorophyll *a* was monitored at site 10, located approximately 3 kilometres north of the Cape Peron outfall, on 16 occasions between December and March 1994 (Cary and D'Adamo, 1995), and these data were used in conjunction with data collected from SMP site 9, located about five kilometers south of the outfall, to assess downstream affects of the outfall (Figure 1). The two sites were sampled on the same day, within approximately two hours of each other (Cary and D'Adamo, 1995).

Table 1. Chlorophyll *a* data collected in Sepia Depression pre- and post-discharge of sewage from the Cape Peron sewage outfall.

Number of sites	Location of site from outfall	Number times site/s sampled	Sampling period	Reference
Pre-discharge				
1	2 km north	34	1977-1981 (monthly)	Chiffings, 1987
4	5 km north/south	6	1981 (March-September)	LeProvost *et al.*, 1981
13	1 km radius	3	1984 (March-May)	LeProvost *et al.*, 1984
Post-discharge				
13	1 km radius	30	1984 (August)-1986 (June)	LeProvost *et al.*, 1986
1	5 km south	23	1991 (March)-1992 (February)	present study
19	5 km north/south	8	1992 (January-May)	Halpern Glick and Maunsell, 1992

3. Results

3.1 Comparison of composite annual cycles for 1977/81 and 1991/92

Mean monthly nutrient, chlorophyll *a* and light attenuation data for Cockburn Sound, Warnbro Sound and Sepia Depression over the two composite annual periods, are presented in Figures 2, 3 and 4. Mean annual data for water quality parameters are shown in Table 2.

Cockburn Sound

Mean monthly nutrient concentrations in Cockburn Sound were consistently lower in 1991/92 compared with 1977/81 (Figure 2a-d). Similarly, mean monthly chlorophyll *a* concentrations and light attenuation coefficients were significantly lower in 1991/92 than 1977/81 (Figures 2f and 2g). When these data are compared on a mean annual basis (Table 2) the chlorophyll *a* concentration in 1991/92 was approximately 30 % lower than in the 1977/81 composite year. Similarly, the mean annual light attenuation coefficient was 15 % lower in 1991/92 compared with 1977/81. Inorganic N:P ratios were significantly higher in 1991/92 than 1977/81 (Figure 2e), and this largely reflects the markedly lower orthophosphate-P concentration in 1991/92 compared with 1977/81 (Figure 2d). In both periods however, the annual mean of these ratios (on a weight basis) was approximately two or less (Table 2), suggesting that nitrogen was the limiting nutrient for plant growth during both periods.

Warnbro Sound

Nutrient concentrations in Warnbro Sound were consistently higher throughout the 1977/81 period compared to 1991/92 (Figure 3a-d), a similar relationship to that found for Cockburn Sound. In contrast, mean monthly chlorophyll *a* concentrations were higher in 1991/92 compared to 1977/81 (Figure 3f); the opposite trend to that found in Cockburn Sound. The mean annual chlorophyll *a* concentration in 1991/92 was about 0.7 $\mu g\ l^{-1}$ (approximately 70 %) higher than in 1977/81 (Table 2). Light attenuation coefficients were not significantly different between the two periods (Figure 3g), however this is largely due to an anomalously high light attenuation coefficient of 0.152 m^{-1} for January 1977/81. If this value is omitted, light attenuation coefficients were generally lower throughout 1991/92 compared with 1977/81. Mean inorganic N:P ratios were generally 5 or less (except in August, 1991/92), and were not significantly different in 1977/81 and 1991/92 (Figure 3e).

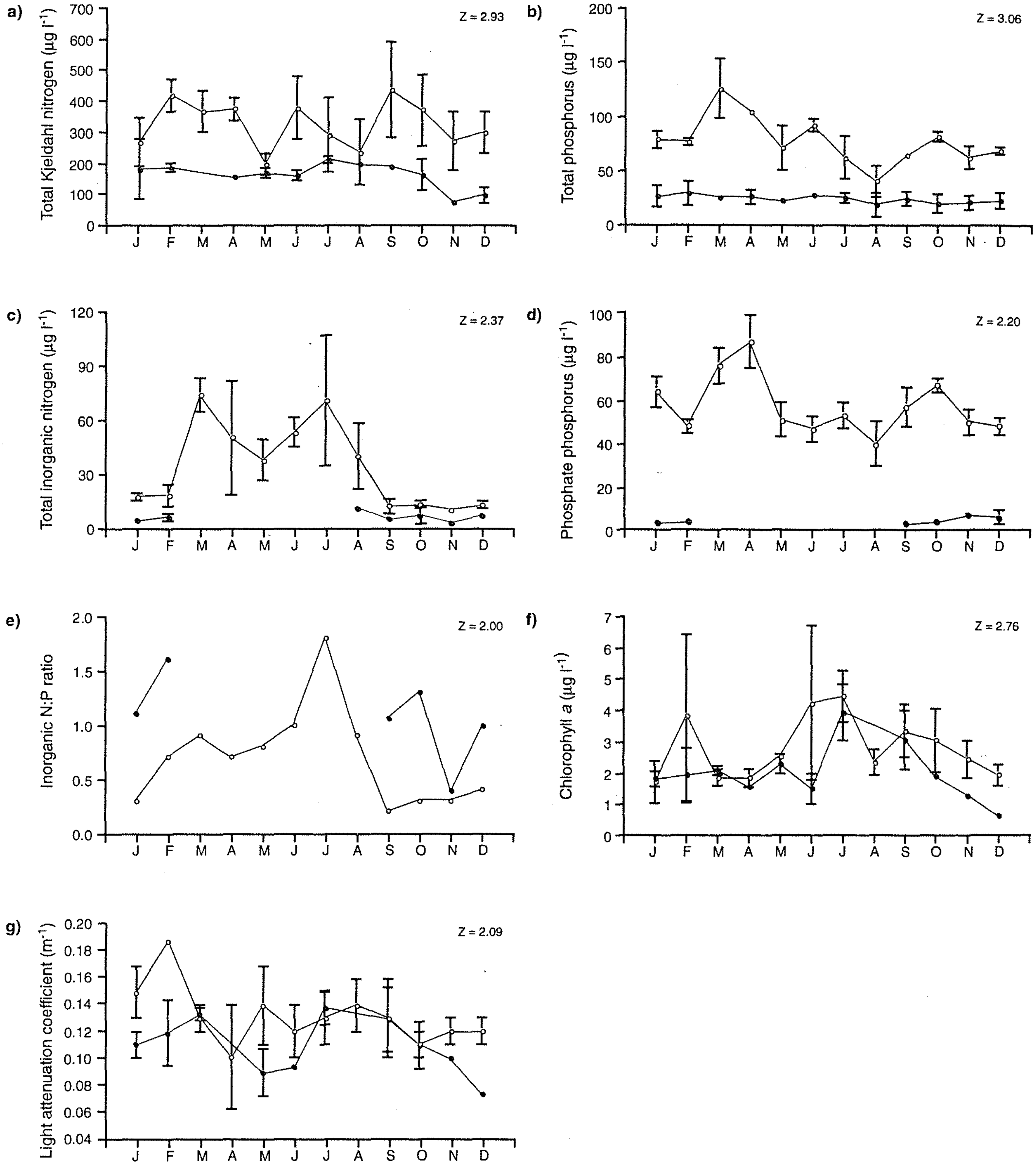

Figure 2. Cockburn Sound mean monthly concentrations of (a) total Kjeldahl nitrogen (b) total phosphorus (c) total inorganic nitrogen (d) orthophosphate phosphorus concentrations; and (e) inorganic N:P ratios (f) chlorophyll *a* concentrations and (g) light attenuation coefficients in 1977/81 (—○—) and 1991/92 (—●—). For the 1977/81 period all data collected from 4 sites (at the same location or close to those used during the 1991/92 survey) on the 34 cruises between July 1977 and June 1981 were used (Chiffings, 1987; Figure 1). For the 1991/92 period all data collected on the 30 cruises over 14 months between January 1991 and March 1992 were used and for January and February the two months of overlap, the value is the mean of both years. Error bars are standard errors. N:P ratios were calculated as the mean of the N:P ratio of each site and sampling date.

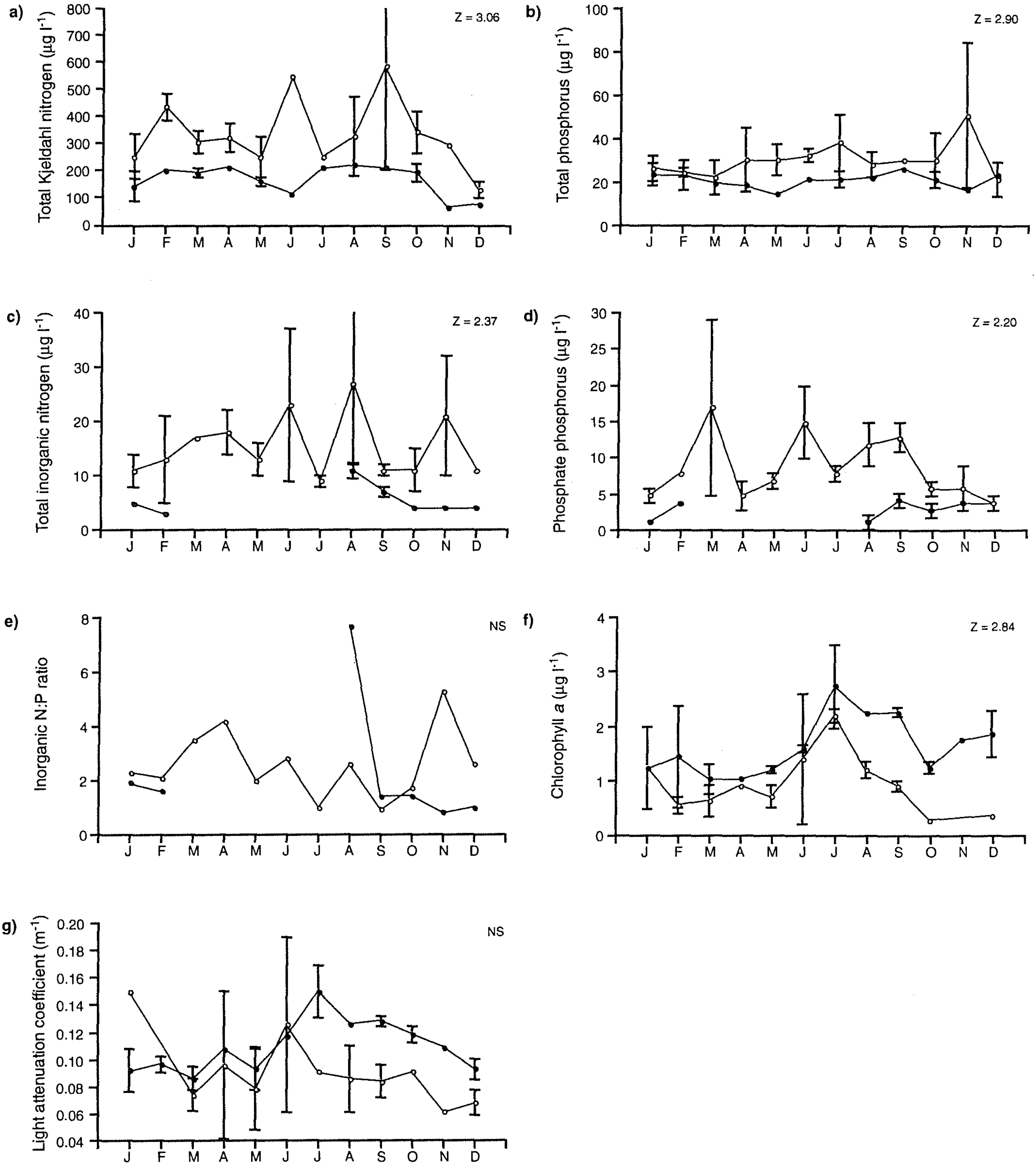

Figure 3. Warnbro Sound mean monthly concentrations of (a) total Kjeldahl nitrogen (b) total phosphorus (c) total inorganic nitrogen (d) orthophosphate phosphorus; and (e) inorganic N:P ratios (f) chlorophyll *a* concentrations and (g) light attenuation coefficients in 1977/81 (——o——) and 1991/92 (——•——). For the 1977/81 period all data collected from site 235 on the 34 cruises between July 1977 and June 1981 were used (Chiffings, 1987; Figure 1). For the 1991/92 period all data collected from the 6 sites in Warnbro Sound on the 30 cruises over 14 months between January 1991 and March 1992 were used and for January and February the two months of overlap, the value is the mean of both years. Error bars are standard errors. N:P ratios were calculated as the mean of the N:P ratio of each site and sampling date.

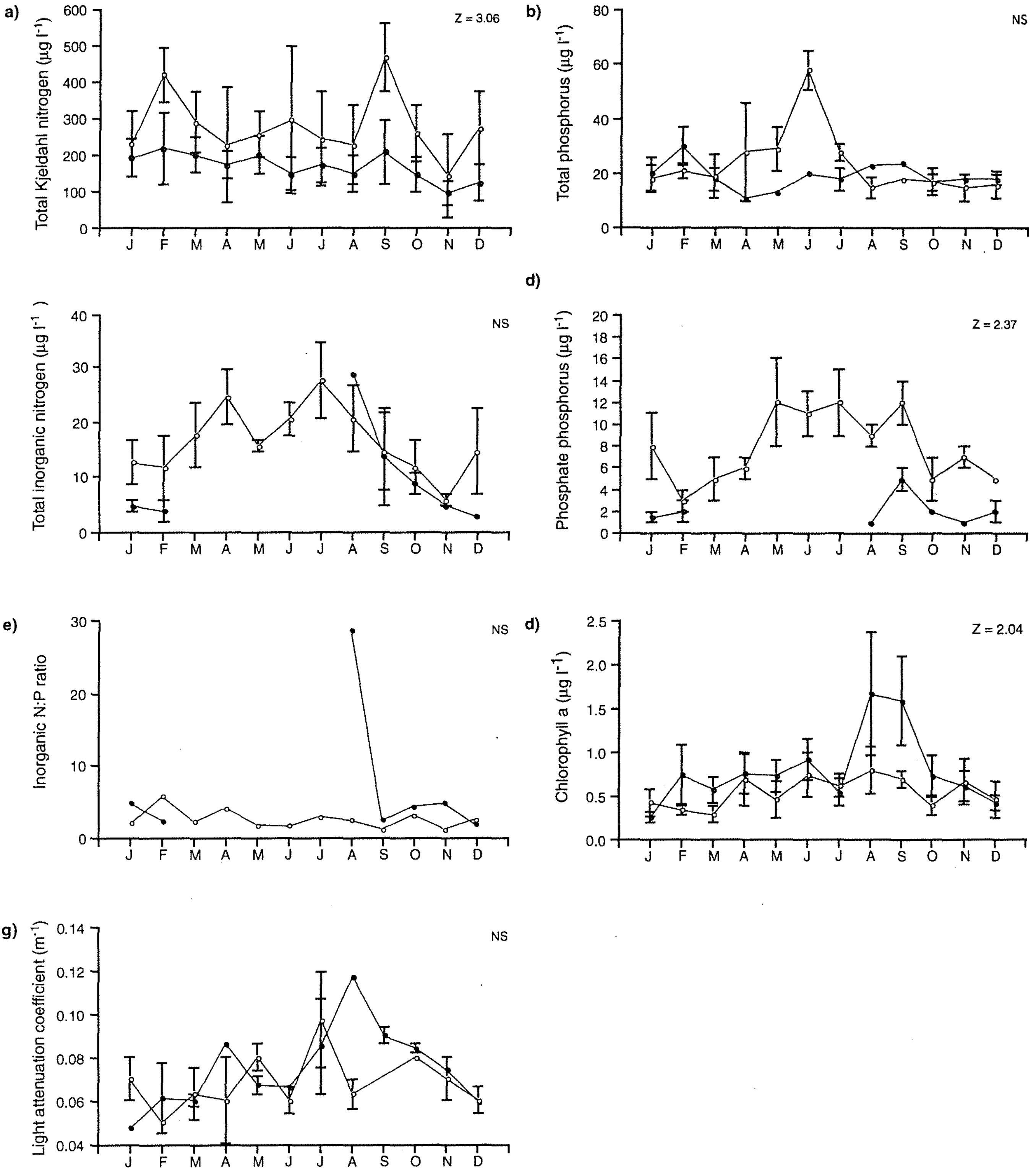

Figure 4. Sepia Depression mean monthly concentrations of (a) total Kjeldahl nitrogen (b) total phosphorus (c) total inorganic nitrogen (d) orthophosphate phosphorus; and (e) inorganic N:P ratios (f) chlorophyll *a* concentrations and (g) light attenuation coefficients in 1977/81 (——o——) and 1991/92 (——•——). For the 1977/81 period all data collected from site 234 on the 34 cruises between July 1977 and June 1981 were used (Chiffings, 1987; Figure 1). For the 1991/92 period all data collected from site 9 in Sepia Depression on the 30 cruises over 14 months between January 1991 and March 1992 were used and for January and February the two months of overlap, the value is the mean of both years. Error bars are standard errors. N:P ratios were calculated as the mean of the N:P ratio of each site and sampling date.

Table 2. Mean annual water quality parameters and nitrogen loads (Martinick *et al.* 1993) calculated over the four year period between 1977 and 1981 and the 14 month period between January 1991 and February 1992. Monitoring sites used to calculate the annual means are shown (see Figure 1). Results of statistical comparisons of composite annual cycles used to derive the mean annual data are also shown. * = ·01< p ≤ ·05; ** = p ≤ ·01; ns = no significant difference; (se) = standard error.

		COCKBURN SOUND			WARNBRO SOUND			SEPIA DEPRESSION		
		1977/81 (Sites 128, 90, 225, 175)	1991/92 (Sites 4, 6, 9, 11)		1977/81 (Site 235)	1991/92 (Sites 2, 3, 4, 6, 7, 8)		1977/81 (Site 234)	1991/92 (Site 9)	
	n	x (se)	x (se)		x (se)	x (se)		x (se)	x (se)	
Chlorophyll *a* ($\mu g\ l^{-1}$)	12	2.8 (0.3)	2.0 (0.3)	**	0.94 (0.17)	1.6 (0.16)	**	0.55 (0.05)	0.8 (0.12)	*
Light attenuation coefficient (m^{-1})	12	0.13 (0.007)	0.11 (0.007)	*	0.09 (0.008)	0.11 (0.006)	ns	0.06 (0.006)	0.07 (0.006)	ns
PO_4-P ($\mu g\ l^{-1}$)	7	55 (4)	6 (1)	**	8 (1)	3 (0.5)	*	7 (1.1)	2 (0.5)	*
TIN ($\mu g\ l^{-1}$)	7	17 (7)	6 (1)	*	15 (2)	5 (1)	*	13 (2)	10 (4)	ns
TP ($\mu g\ l^{-1}$)	12	78 (7)	22 (1)	**	30 (2)	20 (1)	**	24 (3)	19 (1)	ns
TKN ($\mu g\ l^{-1}$)	12	335 (23)	166 (13)	**	334 (38)	169 (16)	**	279 (25)	173 (11)	**
Inorganic N:P ratio	7	0.5 : 1	1:1	*	2:1	2:1	ns	3:1	7:1	ns
Annual N loads (tonnes)		1,800	600		8	17		17	1,500	
Annual inorganic N loads (tonnes)		1,300	600		8	17		16	1,200	
Annual P loads (tonnes)		1,100	40		1	3		5	300	

Sepia Depression

Significant differences in TKN, PO_4-P and chlorophyll *a* concentrations were found between the two periods (Figure 4a-g). TKN and PO_4-P were consistently lower over the annual cycle in 1991/92 compared with 1977/81 (Figure 4a,d), whereas chlorophyll *a* concentrations were higher in 1991/92 compared with 1977/81; a similar trend to that found in Warnbro Sound. The mean annual chlorophyll *a* concentration in 1991/92 was approximately 0.25 $\mu g\ l^{-1}$ higher than in 1977/81, a difference of approximately 45 % (Table 2). Light attenuation coefficients were not significantly different between the two periods (Figure 4g). Mean inorganic N:P ratios were generally 7 or less (except in August, 1991/92), and were not significantly different between 1977/81 and 1991/92 (Figure 4e).

3.2 Long-term trends

Long-term trends in chlorophyll *a* concentrations and light attenuation coefficients of Cockburn Sound, Warnbro Sound and Sepia Depression during summer are shown in Figure 5.

Cockburn Sound

During the late 1970s and early 1980s mean summer chlorophyll *a* concentrations in Cockburn Sound were approximately 2.2 $\mu g\ l^{-1}$ and light attenuation coefficients were approximately 0.13 m^{-1}. There was a marked reduction of over 50 % in both parameters between 1980/81 and 1982/83; the mean chlorophyll *a* concentration falling to 0.8 $\mu g\ l^{-1}$ and light attenuation coefficient to 0.08 m^{-1}. These are the lowest mean summer values recorded since the SMP commenced. Since then, mean chlorophyll *a* concentrations and light attenuation coefficients gradually increased to approximately 1.8 $\mu g\ l^{-1}$ and 0.10 m^{-1} respectively in 1989/90 and have remained relatively constant since then. These current levels are approximately 20 % lower than those found in the late 1970s (Figure 5).

Warnbro Sound

Mean chlorophyll *a* concentrations and light attenuation coefficients in Warnbro Sound have remained relatively constant, at approximately 0.7 $\mu g\ l^{-1}$ and 0.08 m^{-1} respectively, between 1977/78 and 1993/94, except in 1991/92 when the mean chlorophyll *a* concentration was some three times higher (2.1 $\mu g\ l^{-1}$) and associated light attenuation coefficient was 0.10 m^{-1} (Figure 5).

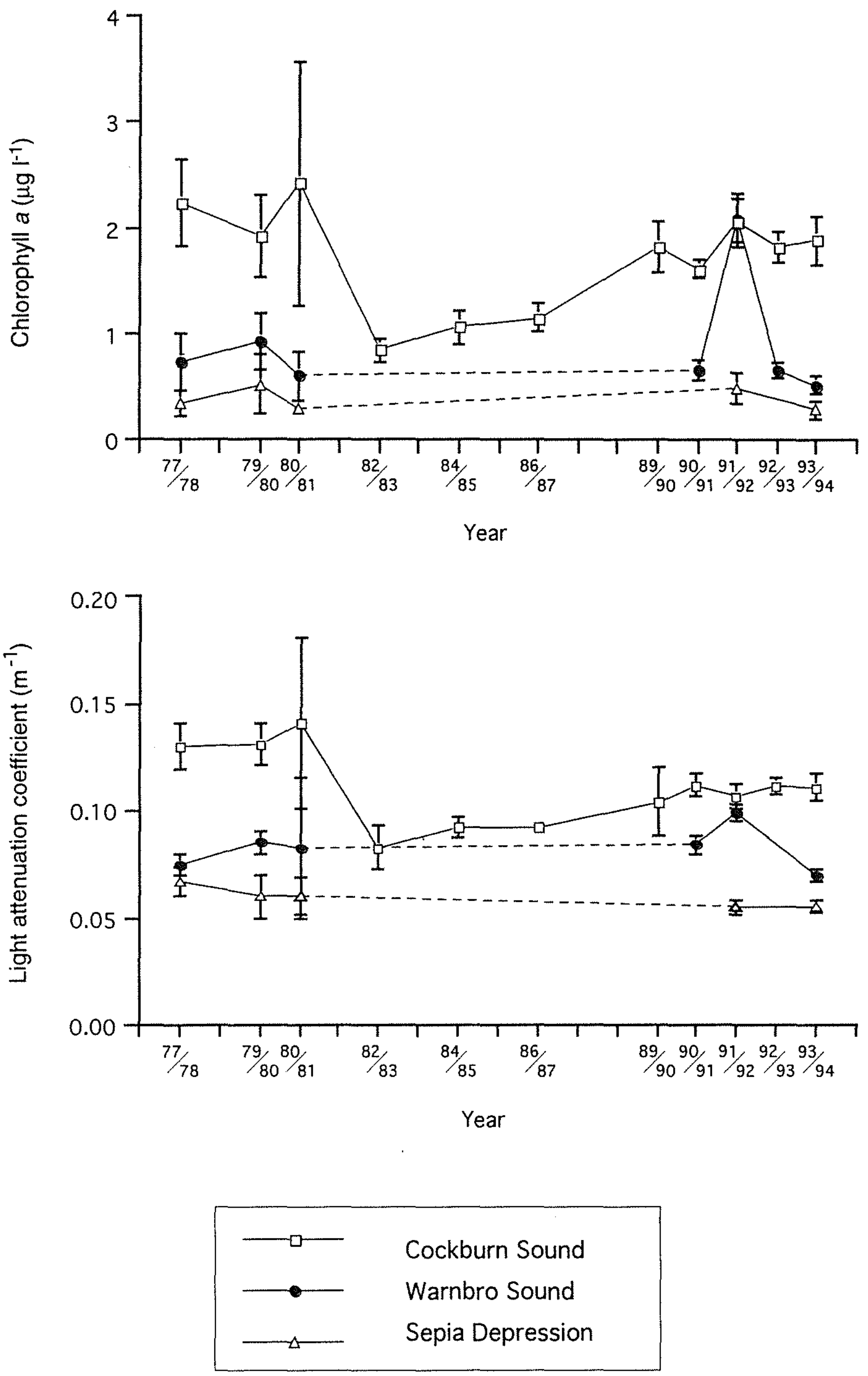

Figure 5. Mean chlorophyll *a* concentrations (a) and light attenuation coefficients (b) in Cockburn Sound, Warnbro Sound and Sepia Depression during the summer months from December to March. For Cockburn Sound 8 sites were sampled in 1977/78 (n=3), 1979/80 (n=4), 1980/81 (n=4), in 1982/83 (n=14), 1984/85 (n=14), 1986/87 (n=14), 1989/90 (n=14), 1990/91 (n=14), 1991/92 (n=16; six of these cruises were data from this study with 4 sites), 1992/93 (n=14) and 1993/94 (n=16; 4 sites). For Warnbro Sound only one site was sampled in 1977/78 (n=3), 1979/80 (n=3) and 1980/81 (n=3), and 6 sites were sampled in 1991 (n=8) and 1991/92 (n=6), and two sites in 1992/93 (n=7; chlorophyll *a* data only) and 1993/94 (n=16). For the one site in Sepia Depression in 1977/78 (n=3), 1979/80 (n=3), 1991/92 (n=6) and 1993/94 (n=16). Error bars are standard errors.

Sepia Depression

Mean chlorophyll *a* concentrations and light attenuation coefficients in Sepia Depression, offshore from Warnbro Sound and approximately 5 km south of the Cape Peron outfall, have remained relatively constant at approximately 0.4 µg l^{-1} and 0.06 m^{-1} respectively, between 1977/78 and 1993/94 (Figure 5). However, a significant ($z = 2.16$; $p<0.04$) increase in chlorophyll *a* concentration was apparent when data from north and south of the outfall (Table 1) were combined (Figure 6a). Using these data, mean annual chlorophyll *a* concentrations increased from 0.53 ± 0.05 µg l^{-1} ($n = 12$) pre-discharge to 0.70 ± 0.1 µg l^{-1} ($n = 12$) post-discharge. The pre-discharge chlorophyll *a* concentration calculated in this way was very similar to the mean concentration measured at site 234 during the 1977/81 pre-discharge period (0.55 ± 0.05 µg l^{-1}). Chlorophyll *a* concentrations were significantly higher ($z = 2.64$; $p<0.009$) at a site approximately 3 km north of the Cape Peron outfall than at a site approximately 5 km south of the outfall, during a 16 week period in the 1993/94 summer (Figure 6b).

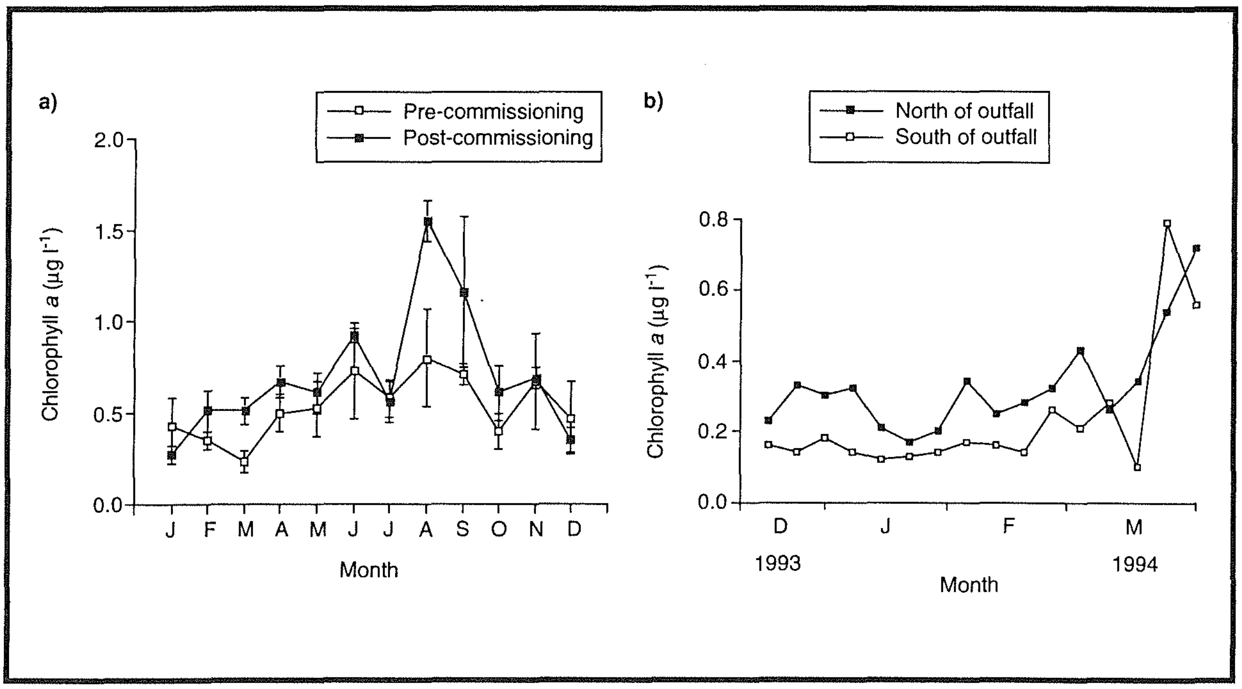

Figure 6. a) Mean monthly chlorophyll *a* concentrations in Sepia Depression pre-discharge (—□—) and post-discharge (—■—) of domestic wastewater from the Cape Peron outfall. b) Chlorophyll *a* concentrations for 16 weekly-cruises over the summer (December - March) 1993/94 at two sites; one site 5 km south (—□—) and one site 3 km north (—■—) of the wastewater outfall.

3.3 Chlorophyll *a* and light attenuation relationship

When mean data were combined from the Cockburn Sound, Warnbro Sound and Sepia Depression SMPs conducted between 1977/78 and 1993/94 (Chiffings, 1979;1987; Chiffings and McComb, 1983; Hillman, 1986; Hillman and Bastyan, 1988; Cary *et al.* 1991; Bastyan and Paling, 1992; Bastyan *et al.* 1994; Cary and D'Adamo, 1995; and the present study) there was a significant correlation ($r=0.94$, $p< 0.04$, $n=22$) between chlorophyll *a* concentrations and light attenuation coefficients (Figure 7).

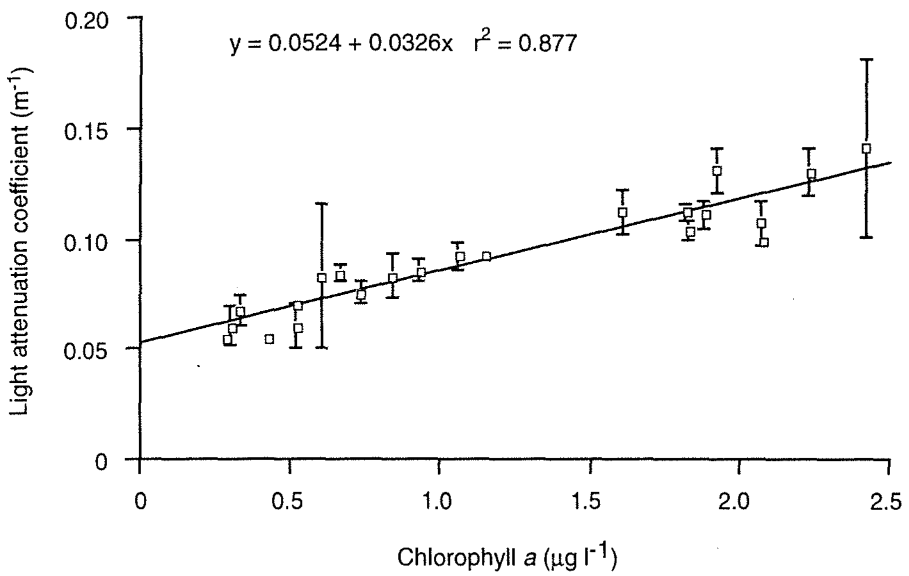

Figure 7. Relationship between mean chlorophyll *a* concentration and light attenuation coefficient derived from all data (see Figure 5) collected during the summer period (December to March) in Cockburn Sound, Warnbro Sound and Sepia Depression. Error bars are standard errors.

4. Discussion

The results of the water quality monitoring programmes evaluated in the present study provide clear evidence that certain water quality parameters of Cockburn Sound, Warnbro Sound and Sepia Depression have changed, to varying degrees, between the late 1970s and the early 1990s. The most obvious change has occurred in ambient nutrient concentrations; in general, levels in all three water bodies over the composite annual cycle in 1991/92 were substantially lower than those found in the late 1970s. The differences in Cockburn Sound can be attributed to substantial reductions in nutrient loading regimes from industrial and domestic sources (Muriale and Cary, 1995). The clearest evidence of this is provided by the PO_4-P data where concentrations fell by more than an order of magnitude over this period in response to a 95 % reduction in annual P load over the same period (Table 2). Inorganic phosphorus appears to be non-limiting to plant growth in these waters, as evidenced by low (<7) inorganic nitrogen to phosphorus ratios, and, as such, will tend to behave more conservatively and therefore reflect loading changes more readily. Inorganic nitrogen, which apparently remains the limiting macro-nutrient, will be readily assimilated by plants and more rapidly removed from the water column.

The cause of the decrease in ambient nutrient concentrations in Warnbro Sound and Sepia Depression is unclear. It is unlikely that the observed differences in ambient nutrient concentrations are analytical or sampling artefacts, as nutrient and chlorophyll *a* determinations were performed by the same laboratory using identical analytical procedures and sampling methodologies and regimes have remained largely consistent over the last 17 years. The one exception to this generality is Warnbro Sound where one site was sampled in the late 1970s compared to six sites in 1991/92. However data from this site can be considered broadly representative of the Sound based on spatial comparisons of the six sites sampled in 1991/92. A major source of interannual variability in nutrient loadings to these waters is most likely related to the intensity and duration of estuarine outflows; however this possibility can be largely discounted, as comparisons of rainfall, which in turn influences estuarine outflow and estuarine-derived nutrient loadings to coastal waters, indicate that rainfall during the winter of 1991 was about 13 % above the long-term average, while three of the four winters of the 1977/81 period had lower than average rainfall. Averaged over the four year period, rainfall at Perth during 1977/81 was 16 % lower than the annual average (Bureau of Meteorology) therefore estuarine-derived nitrogen loadings were probably higher in the 1991/92 period than the 1977/81 period. It should be noted however that rainfall recorded at Perth and flows of the Murray River, which can contribute a significant proportion of the total nitrogen load to Peel Inlet, are not necessarily well-correlated due to its large catchment, and exceptionally high nitrogen loadings to Peel Inlet from this source occurred in 1981 (R J Lukatelich, personal communication).

It is generally accepted that the concentration of inorganic nutrients in oligotrophic coastal marine ecosystems is a poor indicator of nutrient enrichment (ANZECC, 1992) as nutrients can be rapidly assimilated by algae, promoting their growth and leading to elevated algal biomass in the system but low inorganic nutrient concentrations in the water. For example, the production of one μg chlorophyll *a* would require approximately 9 μg of nitrogen, assuming a typical phytoplankton chlorophyll *a* to carbon ratio of 1:50 by weight (Jorgensen *et al.* 1991) and a C:N:P ratio of 41:7.2:1 by weight (Redfield, 1958). In addition, the analytical error for inorganic nitrogen is ±7 μg l^{-1}, approximately 100 % of ambient levels, but the analytical error for chlorophyll *a* is ±0.06 μg l^{-1}, which is between about 3 and 8 % of typical mean levels recorded in the study area (Table 2). Given these constraints, the biomass of primary producers such as phytoplankton are generally considered better indicators of nutrient enrichment than are ambient nutrient concentrations (Anon., 1979; Pearce, 1990). Apart from the usefulness of phytoplankton biomass (measured as chlorophyll *a* concentration) as an 'indicator' of nutrient status, it is also an important water quality parameter in its own right, providing an index of pelagic primary productivity and influencing water clarity (see Figure 7), which in turn influences the viability of important photoautotrophic communities such as seagrasses.

The linear relationship between chlorophyll *a* concentration and light attenuation coefficient that has been established from the SMP data over the last 17 years (Figure 7) is very similar to that derived by Burt *et al.* (1995) for Owen Anchorage during the non-winter period. Other factors such as the concentration of organic and inorganic particulates and dissolved organic substances such as 'Gelbstoff' certainly influence light attenuation, especially in the winter period (Burt *et al.* 1995), but the relationship presented here indicates that chlorophyll *a* is a primary correlate of water clarity in these waters during summer. The linearity of the relationship between phytoplankton chlorophyll *a* and light attenuation coefficient suggests that in these waters chlorophyll *a* is proportional to the other attenuances present, as suggested by Yentsch (1980).

Comparisons of water quality in the embayments and Sepia Depression provide insight into factors that control phytoplankton biomass and hence light attenuation in these waters. Chlorophyll *a* concentrations and water clarity were lower in Cockburn Sound in 1991/92 compared with 1977/81, whereas chlorophyll *a* concentrations were higher in Warnbro Sound and Sepia Depression during the same period (Figures 2f, 3f and 4f). Deeley *et al.* (1995) and Donahue *et al.* (1994) have shown that, in general, above-average rainfall (as occurred in 1991) results in higher than normal nitrogen loads to the Peel-Harvey and Swan-Canning estuaries from their catchments. Discharges from these estuaries can influence the water quality of Sepia Depression and both embayments (Cary *et al.* 1995; D'Adamo *et al.* 1995a,b). Anthropogenic nitrogen loads to the Peel-Harvey Estuary have increased significantly since the 1960s (Deeley, 1995), and blooms of *Nodularia spumigena* (a nitrogen-fixing cyanobacterium) which have occurred most years since the early 1980s are estimated to contribute an extra 2000 tonnes of nitrogen per year to the Estuary in addition to that entering from its catchment (Humphries and Robinson, 1993). *Nodularia* blooms occurred in 1991/92 but in only two of the four years during 1977/81, further supporting the suggestion that nitrogen loading to the coastal waters from estuarine sources would have been higher in the 1991/92 period than during the 1977/81 period, and could partly explain the elevated chlorophyll *a* concentrations in Warnbro Sound and Sepia Depression in the present study (Figure 3f and 4f). Blooms of the cyanobacterium *Trichodesmium* are also common along the Western Australian coast during late summer/autumn (although this is not related to nutrient enrichment), and can dramatically increase the chlorophyll *a* concentration of a waterbody if present (Creagh, 1985). The high spatial variability in the chlorophyll *a* concentration in Cockburn Sound in the summer of 1980/81 (large standard error, Figure 5a) may have been due to a *Trichodesmium* bloom, although there are no data to support this. Another unusual feature of the phytoplankton composition in the study area was the presence of high numbers of *Nodularia spumigena* in Warnbro Sound (Helleren and John, 1995), an estuarine species which does not 'bloom' in marine waterbodies that have salinities above 25 pss (J John, personal communication). Therefore, it is likely the *Nodularia* found in Warnbro Sound originated from the Peel-Harvey Estuary, where it is found in large numbers (Hillman *et al.* 1990), and so was not locally generated. An essentially estuarine diatom species, *Skeletonema costatum*, is the dominant phytoplankton species in the Swan River Estuary (John, 1994) but has been recorded in large numbers in Cockburn Sound during the winter of 1994 (Helleren and John, 1995) and was present through Owen Anchorage, Cockburn Sound and in Sepia Depression to just north of the Cape Peron sewage outfall during a regional survey conducted during August 1991 (Masini and Cousins, 1995). Seasonal patterns in phytoplankton composition, distribution and abundance have been monitored in the study area since 1991 (Cousins, 1991; Cousins and Masini, 1992; Helleren and John, 1995) and this information may assist in the interpretation of subsequent water quality monitoring programmes. The findings discussed above suggest that both estuarine-derived sources of nutrients and externally generated phytoplankton can influence the water quality of the sub-regions in the study area. The relative influence of 'local' and 'regional' scale forcings on water quality is discussed in more detail by Cary *et al.* (1995).

Further insight into the factors that control water quality in the study area can be gained by examination of the water quality of the individual sub-regions.

Cockburn Sound

The data presented here suggest that the water quality of Cockburn Sound, expressed as chlorophyll *a* concentrations and light attenuation coefficients, has improved, albeit slightly, since the late 1970s when the Sound was in its poorest recorded state. This improvement in water quality evident in the 1991/92 composite annual cycle compared with the 1977/81 cycle was associated with a 70 % reduction in annual total nitrogen loading and a 96 % reduction in phosphorus loading to the Sound from industrial and domestic sources (Table 2). The reductions in nutrient loads were also associated with marked reductions in nutrient concentrations. Although rivers contribute to the nutrient loading of Cockburn Sound during winter (Cary *et al.* 1995) the magnitude of these contributions is difficult to quantify and were not included in the annual loading estimates. When interannual comparisons between nutrient loading and chlorophyll *a* are restricted to the summer period only, the nutrient loading : system response relationship is easier to disentangle, as nutrient loadings can be more accurately quantified because river-flow is weak and the nutrient loading is dominated by local (within Cockburn Sound) point source discharges and groundwater inflows (Cary *et al.* 1995).

The relationship between nutrient loading and mean chlorophyll *a* in Cockburn Sound during summer was investigated for 11 summer monitoring programmes (SMPs) from 1977/78 (Figure

8). Estimates of mean daily nitrogen loading during the December to March period were obtained from Murphy (1979), Chiffings (1987), Martinick *et al.* (1993) and Muriale and Cary (1995). Reliable estimates are available for all SMPs except 1979/80 and 1980/81. Estimates of loads from industrial point sources into Cockburn Sound for these two periods were calculated from the data presented by Chiffings (1987) and found to be highly disparate. When the load for 1977/78 was calculated from the annual and monthly load data of Chiffings and compared to the load estimated by Murphy (1979) it was found that the former estimate was about 800 kg d^{-1} or 26 % less than the estimate of Murphy. Murphy's load estimate was based on a comprehensive analysis of effluent composition and flow rates and, there is therefore, good justification for adding 26 % to the mean loads calculated from the data presented by Chiffings (1987) to provide an estimate for 1979/80 and 1980/81 loads. Although high nitrogen loads were also discharged into the Sound from a sewage outfall between 1977/78 and 1984, the nutrient load from this source was not included in the loading estimate on the assumption that the dominant winds during summer would have blown most of the buoyant sewage plume northward out of the Sound (Mills and D'Adamo, 1995).

The poor water quality (expressed as high chlorophyll *a*) of the Sound during the late 1970s was associated with high nitrogen loads (Figure 8), mainly from industrial discharges (Anon., 1979). The marked improvement in water quality of Cockburn Sound between 1980/81 and 1982/83 was associated with a 40 % reduction in nitrogen loading mainly from industrial sources brought about by better industrial practices (Chiffings and McComb, 1983; Martinick *et al.* 1993; Muriale and Cary 1995). The gradual decline in the water quality from the early to late 1980s was associated with a gradual increase in industrial nitrogen loads from point and diffuse (groundwater) sources (Figure 8; Martinick *et al.* 1993; Muriale and Cary, 1995). A relationship between nitrogen load from the dominant industrial source of nitrogen (CSBP and Farmers Ltd)

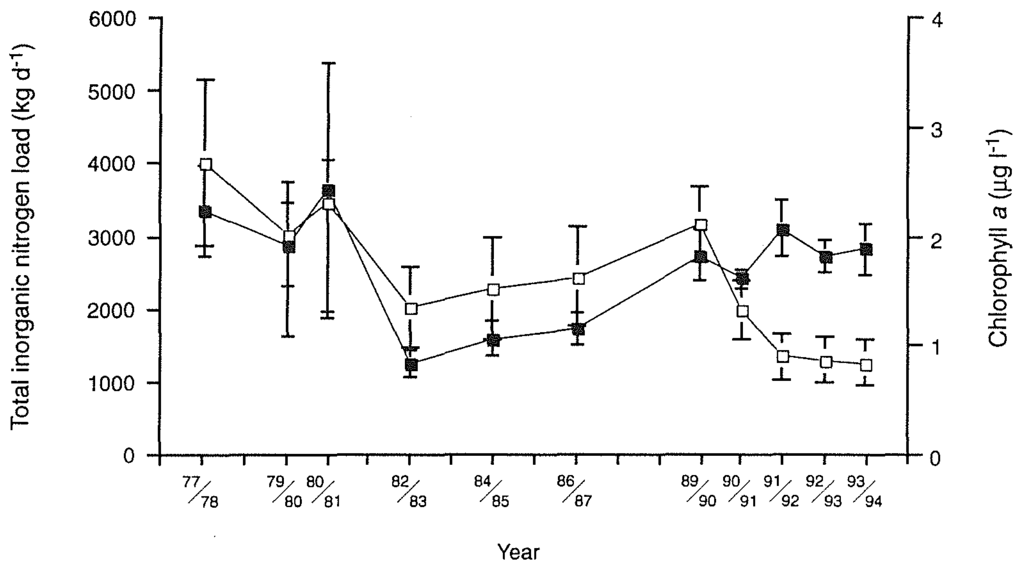

Figure 8. Total inorganic nitrogen loads to Cockburn Sound from all sources (—□—) and chlorophyll *a* concentrations (—■—) between December and March, from the Cockburn Sound summer water quality monitoring programmes between 1977/78 and 1993/94. Data for 1979/80 and 1980/81 are estimates only. For details see Muriale and Cary (1995). Error bars are standard errors except for nitrogen loads in 1979/80 and 1980/81, where the range is presented.

Table 3. The relationship between mean total inorganic nitrogen load (kg d^{-1}) and mean chlorophyll *a* concentrations (µg l^{-1}) in Cockburn Sound using the summer monitoring programmes (SMPs) data between 1977/78 and 1993/94. R = correlation coefficient; p = level of significance; and n = number of SMPs.

SMP periods	R	p	n
1977/78 to 1989/90 (79/80 and 80/81 N load data uncorrected)*	0.78	<0.04	7
1977/78 to 1989/90 (79/80 and 80/81 N load data corrected by +26%)*	0.94	<0.002	7
1977/78 to 1989/90 (excluding 79/80 and 80/81 periods)	0.99	<0.002	5
1977/78 to 1993/94	-	NS	11

*see text for details

and chlorophyll *a* concentration from the early to late 1980s for Cockburn Sound during summer was described by Cary *et al.* (1991). A similar relationship between nitrogen load and chlorophyll *a* also exists when all sources of nitrogen to Cockburn Sound (excluding loads from the sewage outfall as discussed above) and data from the 1977/78, 1979/80 and 1980/81 SMPs are included. This relationship holds regardless of whether the load corrections are applied to the 1979/80 and 1980/81 data (see above) or when these data are excluded (Table 3). However, these relationships do not hold when data from the 4 SMPs conducted between 1990/91 and 1993/94 are included (Table 3). This departure in the loading:response relationship can possibly be explained in several ways; that the original correlation between loadings and mean chlorophyll *a* concentrations was coincidental, that some new ecological process has dominated the nutrient dynamics of Cockburn Sound since 1990, or that current estimates of total nutrient load to the Sound are less accurate than in the past. The probability that the correlation was due to chance is less than 5 % and currently there is no evidence to suggest that key ecological processes of Cockburn Sound have changed significantly since 1990. In relation to the accuracy of loading estimates no new direct discharges have occurred since 1990 (Muriale and Cary, 1995), however estimates of groundwater nitrogen flux to Cockburn Sound (Appleyard, 1990;1994), currently thought to be about 70 % of the total loading, may be much higher due to the considerable error associated with estimating the total mass of nitrogen entering Cockburn Sound from this source (Appleyard, 1994). The uncertainty associated with identifying and accurately quantifying groundwater sources and nitrogen loads is also highlighted by Martinick *et al.* (1993) in their assessment of the quality of the data used to estimate groundwater flux to the Sound and by the 'discovery' in 1990/91 of a major groundwater source to the south of James Point in Kwinana (Figure 1), and subsequent doubling of original nitrogen loading estimates in 1993 after additional investigations (Western Mining Corporation Limited, 1993; 1995).

Warnbro Sound

The composite annual data in Warnbro Sound show that although nutrient concentrations were lower in 1991/92 than 1978/81, chlorophyll *a* concentrations increased (Figure 3). Annual nitrogen loads to Warnbro Sound, although small compared with the loads to Cockburn Sound, have more than doubled since the late 1970s and were estimated to be 17 tonnes in the early 1990s (Table 2). However, is seems unlikely that an increase of about 8-9 tonnes in the annual nitrogen load could have caused the observed high phytoplankton concentration found over the 1991/92 annual cycle. Nitrogen-rich plumes from the Cape Peron sewage outfall have, however, been tracked to the vicinity of Penguin Island under northerly winds (Water Authority, unpublished data) and water from around Penguin Island enters Warnbro Sound under east north-east winds (D'Adamo *et al.* 1995a). Winds that have a significant northerly component are most common between May and October (Steedman and Craig, 1979; 1983) suggesting that during this period at least, contaminants from the Cape Peron outfall influences the water quality of the Shoalwater Islands Marine Park, including Warnbro Sound.

When long-term summer data are examined (Figure 5), the high chlorophyll *a* and reduced light penetration found during the summer of 1991/92 appears anomalous and would suggest that this period was not representative of the current water quality status of Warnbro Sound. Water temperature and global radiation can influence phytoplankton growth but interannual variation in these parameters does not appear to be the cause of this apparent anomaly as data recorded during the 1991/92 and 1977/81 periods were similar to long-term averages (Western Australian Bureau of Meteorology; Hodgkin and Phillips, 1969; Pearce, 1986). Cousins and Masini (1992), in a parallel study of phytoplankton species composition and abundance noted that the winter silicoflagellate bloom in Warnbro Sound reinitiated in the summer of 1991/92 but in subsequent years blooms were largely confined to winter (Helleren and John, 1995). Although silicoflagellates were found in both embayments and Sepia Depression during the winter months, they are not usually abundant after November. If the 1991/92 data point is omitted from the trend analysis on the basis that it was 'atypical' due to the summer silicoflagellate bloom, the long-term summer monitoring data suggest there has been no appreciable change in the chlorophyll *a* concentration or water clarity of Warnbro Sound since the 1970s. This conclusion, however, is based on relatively limited data and many years of data are required to characterise natural variability and therefore to be able to detect declining water quality with surety. Given the documented interannual variability in Warnbro Sound in recent years, the close proximity of a significant, chronic source of nitrogen (ie. the Cape Peron outfall) and potential inputs from ongoing urban expansion in the southern metropolitan area, carefully designed monitoring programmes will be required to detect undesirable trends in water quality and ensure the waters of the Shoalwater Islands Marine Park are protected.

Sepia Depression

Comparisons of the composite annual data indicate that although there was no consistent difference in nutrient concentrations between 1977/81 and 1991/92, phytoplankton abundance (measured as chlorophyll *a* concentration) has increased since the late 1970s. As discussed previously the nutrient loading associated with the above-average rainfall in 1991/92 compared with 1977/81 may have contributed to the elevated chlorophyll *a* levels found in Sepia Depression during the period of river-flow (June-September; Figure 4f) but does little to explain elevation in the non-winter months. Another source of nitrogen of sufficient magnitude to exhibit a 'broad-scale' influence on the water quality of the study area is the Cape Peron sewage outfall discharging into Sepia Depression (Figure 1). Annual anthropogenic nitrogen loading to Sepia Depression has increased nearly 100 fold, from an estimated 17 tonnes in 1977/81 to 1500 tonnes in 1991/92. Until the outfall was relocated to Sepia Depression from Cockburn Sound in 1984, the only major anthropogenic nitrogen loads to Sepia Depression were considered to be from atmospheric deposition (Martinick *et al.* 1993). Nutrient loadings from the Peel-Harvey Estuary were not included at the time as this source was not considered to be significant, whereas it is now largely accepted that the Peel-Harvey outflow can markedly influence the water quality of Sepia Depression, especially during winter (Simpson *et al.* 1993; Cary *et al.* 1995; D'Adamo *et al.* 1995a).

Comparisons of composite annual cycles of chlorophyll *a* concentrations in Sepia Depression show a significant increase in chlorophyll *a* concentrations since the sewage outfall was commissioned in 1984 (Figure 6a), suggesting that the outfall is a contributing factor. This conclusion is strengthened by the conservative approach adopted, which included all available data from upstream and downstream of the outfall instead of only the downstream data, to assess whether changes had occurred.

Data collected north and south of the outfall provide more direct evidence of an effect of the Cape Peron outfall and show significantly higher chlorophyll *a* concentrations approximately 3 kilometres north of the outfall compared with concentrations approximately 5 kilometres south of the outfall (Figure 6b). It is unlikely that the northerly site used in the comparison (site 10) is affected by water from Cockburn Sound as it is located approximately 4 km west of the southern opening to the Sound, and there is a net influx of water from Sepia Depression into the Sound via this opening in summer due to the high occurrence of south southwesterly sea breezes (Binnie and Partners Pty Ltd., 1981; Hearn, 1991; D'Adamo and Mills, 1995). Water currents in Sepia Depression generally run long-shore and are largely wind-driven, and as the prevailing winds during summer have a significant southerly component, water currents will tend to run northward in Sepia Depression at this time of year (D'Adamo and Mills, 1995). An analysis of wind data during the 1993/94 SMP showed that winds preceding each sampling date were predominantly southerly except for sample date 24/3/94, when the dominant wind direction for the preceding 60 hours was from the north. On this occasion the chlorophyll *a* concentration was higher to the south of the outfall. Assuming currents are generally longshore and wind-driven, these data show a consistent elevation in chlorophyll *a*, 3-5 km down-current of the outfall. Recent outfall performance monitoring programmes during May 1995 identified large (up to 10 km^2) patches of elevated NH_4-N and chlorophyll *a* for distances exceeding 4 km down-current (north) of the outfall (Claudius and Nener, 1995).

These data suggest that the zone of influence of the Cape Peron outfall during summer is predominantly to the north, and explains why the summer chlorophyll *a* concentration at site 9, which is located to the south of the outfall, has not changed significantly since the 1970s (Figure 5).

Summary of major findings

- Ecological and analytical considerations indicate that chlorophyll *a* concentrations and light attenuation coefficients are better indicators of the 'health' of oligotrophic marine waters in relation to nutrient pollution than are ambient nutrient concentrations.
- Inorganic nitrogen to phosphorus ratios in the water suggest that nitrogen is the primary macro-nutrient limiting plant growth in these waters, but bio-available phosphorus is also in low concentrations and may limit growth on occasions.
- The water quality of Cockburn Sound between December and March, as indicated by chlorophyll *a* concentrations and light attenuation coefficients, was 'poor' in the late 1970s, showed a marked improvement in the early 1980s but then declined to the point where the current water quality status is only marginally (approximately 20 %) better than in the late 1970s. The initial improvement was related to reductions in total nitrogen loadings from industrial and domestic wastewater. The subsequent decline, until 1990, was related to a gradual increase in industrial nitrogen loads from point and diffuse sources.
- The water quality of Warnbro Sound between December and March, as measured by chlorophyll *a* concentrations and light attenuation coefficients, has remained largely unchanged since the late 1970s probably due to the low nitrogen loadings to the system. The large inter-annual variation in chlorophyll *a* and light attenuation coefficients, as found in 1991/92, underline the importance of understanding natural variation via carefully-designed monitoring programmes when assessing the trophic status of a waterbody.
- Chlorophyll *a* concentrations in Sepia Depression have increased since 1984 suggesting that this increase is primarily in response to the discharge of sewage from the Cape Peron outfall and, in winter, to increased nitrogen loads in the outflow of the Peel-Harvey Estuary.

- Significant relationships were derived between phytoplankton biomass, expressed as chlorophyll *a* concentrations, and light attenuation over the study area and between nitrogen loading and phytoplankton biomass in Cockburn Sound between 1977 and 1990. These relationships indicate that benthic plant communities in the oligotrophic coastal waters of Perth are vulnerable to excessive nitrogen inputs via light limitation.
- The breakdown in the loading:response relationship for Cockburn Sound since 1990, suggests that estimated nitrogen inputs to Cockburn Sound in recent years are inaccurate.

5. Conclusions

Plant growth in Perth's southern coastal waters is nitrogen limited for most of the year and water quality, as measured by chlorophyll *a* concentrations and light attenuation coefficients, is strongly influenced by anthropogenically-derived nitrogen loadings. The water quality of Cockburn Sound responded to reductions in total nitrogen loadings in the early 1980s, and then slowly declined, until 1990, in response to a gradual increase in industrial nitrogen loads from point and diffuse sources, and currently remains in a relatively 'poor' state. The lack of improvement in water quality in response to reductions in nitrogen loads since 1990, suggests that estimated nitrogen loadings to Cockburn Sound since 1990 are inaccurate.

Nitrogen loads to Warnbro Sound are relatively low, and chlorophyll *a* concentrations and light attenuation coefficients during summer on all occasions but one, have not changed significantly since the late 1970s. The high chlorophyll *a* and light attenuation coefficients in the summer of 1991/92 are considered to be 'atypical' and underline the importance of understanding natural variability when assessing the trophic status of a waterbody.

Chlorophyll *a* concentrations in Sepia Depression have increased since 1984 when the Cape Peron sewage outfall was commissioned, and elevated chlorophyll *a* concentrations are now routinely detected down-stream of the outfall suggesting the reduction in water quality in Sepia Depression during non-winter is primarily a response to sewage discharge. In winter, nitrogen loads to these waters from the outflow of the Peel-Harvey Estuary can also reduce water quality.

The established links between nitrogen loading, phytoplankton biomass and light attenuation indicate that benthic plant communities in the oligotrophic coastal waters of Perth are vulnerable to excessive nitrogen inputs via light reduction.

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