Nitrogen over enrichment in subtropical Pearl River estuarine coastal waters: Possible causes and consequences

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ABSTRACT

Hong Kong is surrounded by estuarine, coastal and oceanic waters. In this study, monthly averages over a 10 year time series of salinity, temperature, chlorophyll a (chl a), dissolved oxygen (DO), dissolved inorganic nitrogen (DIN), silicate (SiO4) and orthophosphate (PO4) at three representative stations around Hong Kong were used to examine if excess nitrogen in estuarine influenced waters is due to P limitation. The monthly distribution clearly shows the dominant influence of the seasonal change in river discharge in the Pearl River estuary and adjacent coastal waters. In winter, the river discharge is small and more oceanic waters are dominant and as a result, salinity is high, and chlorophyll and nutrients are low. In summer, when the river discharge is high, salinity decreases and nutrients increase. DIN is very high, reaching 100 μM in the estuary. This indicates over enrichment of nitrogen relative to P and consequently there is an excess of N in coastal waters of Hong Kong. P remains low (~1 μM) and can potentially limit both phytoplankton biomass and N utilization which was demonstrated in field incubation experiments. P limitation would result in excess N being left in the estuarine influenced waters south of Hong Kong. Phosphate concentration is lower in the Pearl River estuary than in many other eutrophied estuaries. Therefore, this relatively low PO4 concentration should be a significant factor limiting a further increase in the magnitude of algal biomass and in the degree of eutrophication in the Pearl River estuary. The export of the excess N offshore into the northern South China Sea may result in an increase in the size of the region that is P limited in summer.

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1. Introduction

The Pearl River estuary is situated on the south coast of Guangdong, China in the subtropical region, and opens to the northern part of the South China Sea. The estuary is partly bounded by Hong Kong to the east. The Pearl River is the second largest river in China next to the Yangtze River and it is the 13th largest river in the world by discharge volume (Lerman, 1981). It stretches for 2214 km and drains an area of 452,000 km² (Zhao, 1990). The Pearl River consists largely of three rivers: the Xi Jiang (West River), Bei Jiang (North River) and the Dong Jiang (East River). The three rivers branch into many small rivers in a network and merge into eight river entrances that empty into the Pearl River estuary and the South China Sea. Four of the river entrances open to the Pearl River estuary between Hong Kong and Macau (Ling Ding Yang in Chinese) (Fig. 1). The yearly average river discharge is 10,524 m³ s⁻¹, with 20% occurring during the dry season in October–March and 80% during the wet season in April–September (calculated from Zhao, 1990). The monthly maximum of the Pearl River discharge is in July, reaching as high as 22,200 m³ s⁻¹. Mean annual rainfall during 1990–1998 was 2480 mm, with 2110 mm (85%) falling during April–September and the maximal rainfall is in June, July and August (calculated from Hong Kong Observatory data, www.hko.gov.hk). The region is subjected to the Southeast Asian monsoons. In winter, the northeast monsoon prevails whereas in summer southwest winds dominate. Therefore, seasonal monsoons and the Pearl River discharge as well as rainfall are the main driving forces in controlling coastal currents, coastal water circulation, mixing and residence time in the Pearl River estuary, and adjacent coastal waters in the region. As a result, the Pearl River estuary and adjacent waters are profoundly influenced by three water regimes: estuarine waters in the Pearl River estuary, oceanic waters from the South China Sea and coastal waters from the China Coastal Current only in winter (Guan, 1994).
The Pearl River Delta region has become one of the fastest developing regions in China in recent years. The region, including Hong Kong, has 45.5 million inhabitants (Enright and Scott, 2006). The Pearl River receives a high load of anthropogenic nutrients from increased activities in agriculture (Neller and Lam, 1994), fish diking farming (Ruddle and Zhong, 1988) and sewage effluent (Hills et al., 1998) due to the population increase and economic development in the expanded Pearl River delta region (Enright and Scott, 2006). The industrial and domestic waste waters development in the expanded Pearl River delta region (Enright and Scott, 2006). The Pearl River-northern South China Sea system and the Mississippi River-northern Gulf of Mexico system: both are subtropical large influenced waters and also to test the hypothesis that P limitation would result in excess N in the estuarine coastal waters in the
region, using mixing experiments between freshwater and seawater.

2. Materials and methods

Since Hong Kong is located in the eastern part of the Pearl River estuary, coastal waters of Hong Kong and other neighboring cities are heavily influenced by the Pearl River estuary. There are over 80 stations which the Environmental Protection Department (EPD) of the Hong Kong Government uses to maintain a comprehensive sampling program to monitor water quality in the territorial waters (EPD, 1999). In this study, three stations representing different geographical locations in Hong Kong were selected: NM5 is located in the Pearl River estuary, SM18 is in southern Hong Kong and MM16 in northeastern waters (Fig. 1). Therefore, these three stations are representative of the estuarine influence (NM5), the estuarine coastal plume (SM18) and oceanic conditions (MM16). Monthly or bi-monthly data were collected by the EPD on the government vessel ‘Catherine Lam’. A SEACAT19 CTD was used to take vertical profiles of salinity, temperature and other parameters. Water samples were taken using the CTD rosette at three depths: surface, middle and bottom (1 m above the bottom) for water quality parameters, from which chl \( a \) and nutrients (\( \text{NO}_3, \text{NO}_2, \text{NH}_4, \text{PO}_4 \) and \( \text{SiO}_4 \)) were used in this study. If the water depth was less than 4–5 m, the middle depth was omitted. Data for each sampling month during 1999–2000 were grouped together from 11 years and were averaged to obtain monthly averages of salinity, temperature, chl \( a \), \( \text{DIN} \), \( \text{PO}_4 \), N:P ratios, and N:Si ratios. For NM5, data in even months were only from 2 years: 1999 and 2000. The correlation coefficient, \( r \), in Tables 1 and 2, was calculated using Microsoft\textsuperscript{®} Excel 97. The statistical test for significance is based on critical values of the correlation coefficient, \( r \) (Zar, 1999).

2.1. Field mixture experiments

At a station (B6) in the middle of the estuary (Fig. 1) on June 16, 1998, a surface water sample was taken with a clean plastic bucket and a sample at 7 m below the halocline was taken with a

<table>
<thead>
<tr>
<th>Stations</th>
<th>Temp.</th>
<th>Chl ( a )</th>
<th>( \text{NO}_3 )</th>
<th>( \text{SiO}_4 )</th>
<th>( \text{PO}_4 )</th>
<th>( \text{NH}_4 )</th>
<th>N:P</th>
<th>N:Si</th>
</tr>
</thead>
<tbody>
<tr>
<td>NM5 (n = 88)</td>
<td>-0.70*</td>
<td>-0.06</td>
<td>-0.87*</td>
<td>-0.91*</td>
<td>-0.16</td>
<td>0.25*</td>
<td>-0.50*</td>
<td>0.02</td>
</tr>
<tr>
<td>SM18 (n = 74)</td>
<td>-0.60*</td>
<td>-0.68*</td>
<td>0.89*</td>
<td>-0.81*</td>
<td>-0.06</td>
<td>-0.06</td>
<td>-0.51*</td>
<td>-0.51*</td>
</tr>
<tr>
<td>MM16 (n = 47)</td>
<td>-0.40*</td>
<td>-0.31*</td>
<td>-0.36*</td>
<td>-0.26</td>
<td>0.04</td>
<td>-0.13</td>
<td>-0.19</td>
<td>-0.18</td>
</tr>
</tbody>
</table>

The degrees of freedom, \( n-2 \), are equal to the sampling times 2. The bold values with * indicate that \( p < 0.05 \), which is considered to be statistically significant based on the critical values for \( r \) by Zar (1999).

<table>
<thead>
<tr>
<th>Stations</th>
<th>Secchi depth</th>
<th>Temp.</th>
<th>( \text{NO}_3 )</th>
<th>( \text{SiO}_4 )</th>
<th>( \text{PO}_4 )</th>
<th>( \text{NH}_4 )</th>
<th>N:P</th>
<th>N:Si</th>
</tr>
</thead>
<tbody>
<tr>
<td>NM5 (n = 88)</td>
<td>0.21*</td>
<td>0.04</td>
<td>0.05</td>
<td>-0.17</td>
<td>-0.37*</td>
<td>-0.14</td>
<td>0.40*</td>
<td>0.67*</td>
</tr>
<tr>
<td>SM18 (n = 74)</td>
<td>-0.09</td>
<td>0.46*</td>
<td>0.57*</td>
<td>0.53*</td>
<td>-0.12</td>
<td>-0.16</td>
<td>0.40*</td>
<td>0.26*</td>
</tr>
<tr>
<td>MM16 (n = 47)</td>
<td>-0.12</td>
<td>-0.19</td>
<td>0.38*</td>
<td>0.13</td>
<td>-0.09</td>
<td>0.00</td>
<td>0.31*</td>
<td>0.28</td>
</tr>
</tbody>
</table>

The degrees of freedom, \( n \), are equal to the sampling times 2. The bold values with * indicate that \( p < 0.05 \), which is considered to be statistically significant based on the critical values for \( r \) by Zar (1999).

![Fig. 1. Map showing the Pearl River estuary and sampling stations.](image-url)
Niskin bottle. The salinity of the surface sample and seawater sample was 5 and 30, respectively. They were directly mixed at different ratios of surface to deep seawater: 100–0%, 75–25%, 50–50%, 25–75% and 0–100%, which will be referred to as FW100–SW0, FW75–SW25, FW50–SW50, FW25–SW75 and FW0–SW100, respectively. The mixtures were transferred into 2-L transparent glass bottles in duplicate and placed in a plastic box for incubation. The box was flushed frequently with surface water to maintain a temperature similar to the surface seawater. The temperature of the incubator was monitored with a thermometer. Overheating was not a problem since air temperature was lower than the water temperature. The samples were incubated in field to sunlight and hence, phytoplankton might have experienced some photoinhibition, which could have slowed down phytoplankton growth and uptake of nutrients. This would not affect which nutrient is the most limiting unless there is a large differential uptake of different nutrients by phytoplankton under photoinhibition. During the 4-day incubation, water samples were taken twice a day, once in the morning and once in the evening. In vivo fluorescence was measured with a fluorometer (Turner Designs AU10) on board. Other analyses for chl a and nutrients (NO3+NO2, NH4, PO4 and SiO4) were conducted later in the laboratory. The chl a samples were extracted with 90% acetone in darkness at 4 °C for 24 h and chl a analysis was measured by in vitro fluorescence (Parsons et al., 1984) measured on a Hitachi fluorescence spectrometer. For the EPD data, chl a was extracted using acetone and measured spectrophotometrically (APHA, 1995; c.f. EPD, 1999).

2.2. Nutrient analysis

The analytical methods followed the standard methods for the examination of water and wastewater and were given in the Marine Water Quality Report, Hong Kong (EPD, 1999). Our nutrient analysis was similar to EPD’s methods. Basically, NO3 and NH4 were analyzed by the Cu–Cd column reduction method (APHA, 1995; c.f. EPD, 1999) and the indophenol blue color formation, respectively (EPD, 1999). The low detection limits were 0.05 μM for both NO3 and NH4. Soluble PO4 (orthophosphate) was measured using the ascorbic acid method (EPD, 1999) and SiO4 was analyzed using molybdate, oxalic acid and a reducing reagent (EPD, 1999) and the low detection limit were 0.05 and 1 μM, respectively. The water samples for nutrients were not filtered and as a result, PO4 and NH4 were higher in an unfiltered sample than in the filtered sample (K. Yin, unpublished data). NO3 and SiO4 are much less affected by filtration vs no filtration based on our measurements (K. Yin, et al. unpublished data). There are implications for our assessment of P limitation, as the EPD data might have given higher P concentrations than actual available P to phytoplankton. Therefore, our assessment and conclusions are conservative since P can be more frequently limiting than the reported EPD concentrations indicate. No filtration of samples has less implications for other nutrients and will not affect our conclusions in this study. Molar concentrations of NO3+NO2+NH4 (dissolved inorganic nitrogen= DIN) and PO4 and SiO4 were used to calculate atomic ratios of N:P and N:Si.

3. Results

There was spatial and temporal variation in salinity and temperature in Hong Kong waters. Surface salinity changed between >30 in January and 5 in July at NM5, and at SM18 the change in surface salinity was similar (Fig. 2), but surface salinity usually remained >20 in June. At MM16, the fluctuation in salinity was much smaller as surface salinity only decreased to 27 in summer. Salinity in the bottom water decreased with the surface salinity at NM5 except for April. At SM18 and MM16, however, the bottom salinity increased in summer when the surface salinity decreased. Temperature at the surface showed a strong seasonal change between about 27–29 °C in summer and 16–18 °C in winter at all stations (Fig. 2). When the surface temperature increased during summer, the bottom temperature decreased at SM18 and MM16 (Fig. 2). The onset of the water column stratification (the halocline formation) occurred in March
at NM5 and progressed to MM16 in May. In contrast, thermal stratification started earlier at MM16 and later at NM5 (Fig. 2).

The chl a concentration at NM5 was about 3–5 µg L⁻¹ during December–May except for March, and remained < 3 µg L⁻¹ during June–November (Fig. 3). The highest monthly chl a concentration at NM5 in May was < 7 µg L⁻¹. At SM18, the peak chl a concentration was during June (ca. 11 µg L⁻¹), one month later than at NM5 and decreased to 4 µg L⁻¹ in August, remaining < 3 µg L⁻¹ during other months. The chl a concentration at MM16 was low (< 2 µg L⁻¹) throughout the year (Fig. 3). For the bottom, chl a concentrations at NM5 increased at the same time as chl a at the surface, and only slightly at SM18 (Fig. 3). Chl a at MM16 was even slightly higher at the bottom than at the surface (Fig. 3).

Surface dissolved oxygen (DO) decreased in summer at NM5 and showed small temporal changes at SM18 and MM16 (Fig. 3). There was no difference between bottom and surface DO during winter months. Bottom DO started to decrease in May, decreasing to 4 mg L⁻¹ at NM5 during July–September with the lowest (2.2 mg L⁻¹) in June. The lowest bottom DO was about 4 mg L⁻¹ in August at SM18 and in July at MM16 (Fig. 3).

There was a seasonal change in the surface DIN concentration at NM5 with lows in winter (~20 µM) and peaks (~100 µM) in
summer (June and August) (Fig. 4). Seasonal changes in DIN occurred with the peak of 30 μM and the lows <5 μM at SM18. The seasonal changes in DIN were lacking at MM16. DIN concentrations at the surface and bottom at MM16 were very low, <5 μM all year round, except for July at the bottom (Fig. 4). The seasonal change in SiO₄ was very similar to DIN at NM5 and SM18. SiO₄ concentrations were generally similar to DIN at these two stations, but were higher than DIN at the surface at MM16 (Fig. 4). PO₄ concentrations were around 1 μM at NM5 (Fig. 5) except for August. At SM18 and MM16, PO₄ concentrations were <0.6 μM all year round. In contrast to DIN and SiO₄, PO₄ showed no particular seasonal variations at both the surface and bottom (Fig. 5), probably a result of not filtering water samples.

There were strong seasonal variations in the N:P ratio at NM5 and SM18 (Fig. 6). At NM5, the N:P ratio at the surface was near 32:1 during January–March and increased >48:1 between April and September and was around 20:1 during October–December. The N:P ratio exceeded 80:1 at SM18 during June and August. However, at MM16, N:P ratios were <16:1 all year. N:Si ratios were >1:1 during January–May and close to 1:1 during the remainder of the year at NM5. N:Si ratios were <1:1 at SM18, although there was a seasonal change with peaks during June and August, and lows in other months. N:Si ratios at the surface were very low (<0.4:1) at MM16 (Fig. 6).

Most of the fluctuations in surface NO₃ and SiO₄ were significantly negatively related with variations in salinity at NM5 and SM18 (Fig. 7). At NM5, 77% and 83% of the variations in NO₃ and SiO₄, respectively, were accounted for by the variation in salinity, and at SM18, 80% and 65% was accounted for. In contrast, there was no significant correlation between salinity and NO₃ or SiO₄ at MM16 (Fig. 7).

The incubation of the different mixtures of freshwater and seawater showed that PO₄ disappeared to <0.1 μM after <30 h in all the mixtures except for the freshwater mixture (FW100–SW0), and NO₃ and SiO₄ disappeared after about 54 h in the seawater sample (FW0–SW100). However, both NO₃ and SiO₄ could not be depleted in the intermediate mixtures after about 54 h (Fig. 8).

4. Discussion

The time series of nutrients clearly showed that there was high nitrogen in the Pearl River estuary all year around and N was in excess relative to P in southern waters of Hong Kong. Chl a concentrations were not as high as expected considering the high...
N concentrations, since 1 µM N can be roughly converted into 1 µg L⁻¹ chl a during phytoplankton growth (Yin et al., 1996). There are different limiting factors which are responsible for the differences in chl a and nutrients among the three stations.

The Pearl River estuarine plume flows eastwards and covers Hong Kong coastal waters during the wet summer season (Mao et al., 2004). Typically, as the estuarine/coastal plume moves downstream, salinity at the surface increases, light penetration improves, chl a concentration increases and nutrients decrease. It appears that low chl a in winter was mainly due to the lack of stratification due to frequent wind mixing. In summer, low chl a in the Pearl River estuary suggested light limitation as the estuary is usually turbid during the summer due to high freshwater discharge. The southern waters of Hong Kong were often at the inner edge of the estuarine/coastal front where chl a increases to a maximum, forming the regional chl a maximum. The regional maximum of chl a was observed to move from the estuary entrance in June, eastwards to the waters south of Hong Kong Island in July 1998 (Yin et al., 2000). During July 1999, the chl a maximum was observed in the eastern waters of Hong Kong as the coastal plume covered the entire southern waters of Hong Kong (Yin et al., 2001). In the region of the chl a maximum, phosphorus was depleted, but there was excess of N. For example, nutrient addition experiments on water samples collected from the region of the chl a maximum (N:P ratio > 160:1) showed the largest response to P additions, whereas beyond the maximum, N/P ratios were < 10:1 and the largest increase in chl a fluorescence was due to N additions (Yin et al., 2001). Therefore, based on the previous observations and the time series of N:P ratios in this study, low P appears to be responsible for the excess N concentrations in the southern waters. In the eastern waters, potential N limitation is more dominant.

There is a two-layer flow estuarine circulation in the Pearl River estuary with the estuarine plume flowing seaward at the surface and the salt wedge moving upstream at the bottom (Mao et al., 2004). Upward mixing by entrainment occurs between the two layers and thus the estuarine plume is basically different mixtures containing different portions of seawater as it moves downstream. When a mixture moves downstream in the

![Fig. 7](image-url). The mixing diagrams of NO₃ vs salinity, and SiO₄ vs salinity (all individual data over 10 years for all seasons).

![Fig. 8](image-url). The mixing experiment in which freshwater at a station in the middle of the estuary is mixed with seawater from below the halocline in different proportions: 100% freshwater (100/0), 75% freshwater vs 25% seawater (75/25), 50% freshwater vs 50% seawater (50/50), 25% freshwater vs 75% seawater (25/75) and seawater only (0/100).
estuarine plume, nutrients will be gradually utilized. When P disappears first, it may limit N utilization, which would result in excess N. The monthly averaged distribution of nutrients and the mixing diagram between salinity and nutrients clearly shows the dominant influence (a decrease in salinity) of the Pearl River estuary on southern waters of Hong Kong during the wet season with excess N and high N/P ratios in the time series. The mixture experiment was to simulate how nutrients would disappear in the various mixtures of estuarine/coastal plume waters. The results from the mixture experiment demonstrated that NO3 in the mixtures of freshwater portion >25% (salinity <22.5) could not be depleted during 3 days (Fig. 8). It is worth pointing out that this result was a net outcome after P regeneration or recycling processes had contributed to the effect of P on N uptake. As a result, excess NO3 would be transported eastwards to Hong Kong or further downstream and offshore.

High nutrient loading from rivers has been found in many estuaries in North America and northern Europe (Howarth et al., 1996) and it has become an increasing concern in causing eutrophication (Jickells, 1998; NRC, 2000). Nitrogen in the Pearl River is certainly high compared with other rivers such as the Amazon River (10–15 μM) (Edmond et al., 1981) emptying into the South Atlantic Ocean and the Sacramento and San Joaquin Rivers (<25 μM) discharging into San Francisco Bay (Schemel and Hager, 1986) and even the largest chinese river, the Yangtze River (Shen, 1993). NO3 in the Pearl River is comparable to that in the Mississippi River 100–120 μM (Turner et al., 1998) and Chesapeake Bay where NO3 in the freshwater region ranged between 70 and 120 μM (Fisher et al., 1988; Ward and Twilley, 1986), although lower than the Rhine River (270 μM, Schaub and Gieskes, 1991). PO4 concentration was ~1 μM in the Pearl River estuary, which was similar to many measurements made in the Pearl River and historical data (Cai et al., 2004). Since the samples were not filtered, this value should be approximately 0.5–0.7 μM if filtered (K. Yin, unpublished data). This range of concentrations is similar to the Amazon River (Edmond et al., 1981), the Yangtze River (Shen, 1993) and Chesapeake Bay (Fisher et al., 1992). PO4 in the Pearl River estuary is definitely much lower than in the Rhine River (0.35 PO4-P mg l−1 (11 μM) (Schaub and Gieskes, 1991), and lower than the Mississippi and Sacramento Rivers where PO4 concentrations are 3 μM (Dortch and Whittle, 1992; Rabalais et al., 1996), and 4 μM (Schemel and Hager, 1986). Serious eutrophication effects like hypoxia, or anoxia have been reported in the northern Gulf of Mexico (Rabalais et al., 1992, 1994; Malalikoff, 1998) and Chesapeake Bay (Breitbart, 1990; Malone et al., 1988).

Since the South China Sea is an oligotrophic ocean, concentrations of nutrients in the estuarine influenced waters are mainly determined by freshwater proportions during mixing. Thus, the mixing between a nutrient and salinity can indicate whether the freshwater end member concentration has changed. If the concentration of NO3 and SiO4 from the freshwater has increased significantly over the years, the concentration in the freshwater–seawater mixture at the same salinity should have increased, and the mixing diagram would result in a deviation from the conservative mixing line. Thus, the significant correlation between salinity and NO3 or SiO4 (Table 1) suggest that interannual variability in concentrations of NO3 and SiO4 from the freshwater source have not changed significantly during the 10 year period during which the data were collected. Lack of significant correlation for NO3 and SiO4 at MM16 (Table 1) suggests that the estuarine influence is not a dominant source of NO3 and SiO4 for waters at MM16.

Phosphorus limitation may play an important role in controlling phytoplankton biomass in the region. Based on N concentrations in the Pearl River estuarine/coastal plume, chl a should have reached much higher concentrations than those observed. For example, at SM18, the average N concentration is 30 μM and chl a was 10 μg l−1. Using 0.9 μg chl a 1−1/μM N as a conversion factor (Yin et al., 1996), 30 μM N would produce 27 μg l−1 chl a. Including the existing 10 μg chl a 1−1, chl a should be as high as 47 μg l−1 if all of the NO3 was used up. However, PO4 is usually <1 μM in the southern waters of Hong Kong at SM18 and would produce <16 μg l−1 chl a assuming average phytoplankton 16N: 1P ratio and the N-chl a conversion factor of 0.9 (1 μM N to 0.9 μg chl a 1−1). Statistical analysis shows that chl a was significantly and positively correlated with NO3 and SiO4 in the southern waters of Hong Kong (Table 2). Normally, chl a should be inversely related to nutrients when they are in the same water body such as a lake or a poorly flushed bay. A positive relation at a station indicates a change of water masses during different sampling times. This suggests that chl a in the southern waters of Hong Kong varies with the spatial variation of the coastal plume that changes with seasons, as suggested by the significant correlation between salinity and chl a (Table 1).

During the last 20 years, anthropogenic phosphorus enrichment did not appear to result in an increase in ambient phosphate since P is low in the Pearl River estuary and the coastal plume (Yin et al., 2000, 2001) in contrast to the Mississippi River and Gulf of Mexico where the phosphorus concentration is high (3–5 μM, Dortch et al., 1992). The reason for low P in the Pearl River needs further study. The phosphate concentration in the surface South China Sea is also low (<0.5 μM, SCSIO, 1985). Dissolved PO4 in 1980 in the Pearl River estuary (SCSIO, 1985) was generally <1 μM in summer which is similar to present concentrations. Therefore, low phosphorus is believed to be a factor limiting the utilization of excess N and a further increase in chl a in the Pearl River estuary and coastal waters of Hong Kong.

In summary, there is nitrogen over enrichment in the Pearl River estuary and adjacent coastal waters, but low phosphorus appears to keep severe N-based eutrophication from occurring. Although N is usually considered to be the most limiting nutrient in coastal marine waters, it is common that phosphorus is more limiting for primary production than nitrogen in nearshore tropical marine systems (Howarth et al., 1995). However, away from the shore, oligotrophic tropical seas may be N limited (Corredor et al., 1999). This switch from the P-limited Pearl River estuary to the N-limited South China Sea system raises a serious question of what happens to the excess N. When excess nitrogen is transported offshore into the South China Sea, it may fuel primary productivity and drive a region in the northern South China Sea into P limitation since the northern South China Sea is reported to be N limited (Wu et al., 2003). The spatial scale of the region depends on how large the Pearl River estuarine coastal plume covers or influences, and the degree of N limitation in the northern South China Sea. The increase in primary productivity may result in more phytoplankton sinking to the bottom layer and consuming more DO. The bottom water moves onshore at the bottom during summer as a compensation for the surface outflow of the coastal plume and also due to upwelling induced by the southwest monsoon and thus these lower oxygen bottom waters would eventually influence the DO level nearshore. So far, the level of DO in the Pearl River estuary and coastal waters has not shown an apparent decreasing trend (Yin et al., 2004). Although N may be limiting in the South China Sea, P concentrations are also very low in these oligotrophic waters. A small amount of N could tip the balance of the N:P ratio to P limitation and therefore excess N may be used up over a large area of the northern South China Sea. Thus phytoplankton production and subsequent DO consumption in the bottom layer would be widely distributed so that they would not cause severe DO environmental problems. The fate of excess N in the northern South China Sea and how it is related to coastal environments deserves further study.
Acknowledgments

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