

## Intertidal marsh as a source of dissolved inorganic carbon and a sink of nitrate in the Satilla River–estuarine complex in the southeastern U.S.

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### Abstract

Total dissolved inorganic carbon (DIC), total alkalinity (TA), pH, and nitrate + nitrite ( $\text{NO}_x$ ) data collected during the summer of 1996 in the Satilla River estuary in the southeastern U.S. were used to assess fluxes of DIC and  $\text{NO}_x$  between intertidal marshes and estuarine waters and to model system  $\text{NO}_x$  dynamics. Nitrate and nitrite are produced in the low-salinity portion of the estuary. The intertidal marshes are sites of intensive respiration that export DIC to the estuary and remove  $\text{NO}_x$ . An integrated view is presented on the nitrification and denitrification processes in the marsh/estuarine complex and their relationship to  $\text{CO}_2$  generation rates. The distribution of  $\text{NO}_x$  in the marsh and estuarine waters indicates that all  $\text{NO}_x$  generated in the marsh–estuary system is removed in the intertidal marshes, most likely via denitrification. Model analysis of  $\text{NO}_x$  and river flow data for three seasons indicates that  $\text{NO}_x$  distribution in the estuarine water is also determined by river flow rates. Export fluxes of  $\text{NO}_x$  to the coastal ocean are insignificant in all seasons when compared to  $\text{NO}_x$  production rates in the entire system; however, they are significantly higher than  $\text{NO}_x$  inputs from the river end member in October 1996. Although only a small fraction (~10%) of DIC generated in the marshes is exported to the coastal sea and around 90% is lost to the atmosphere, it represents a nearly threefold increase in riverine DIC flux to the ocean.

The concentrations of dissolved inorganic carbon (DIC), oxygen ( $\text{O}_2$ ), and inorganic nutrients in estuaries and their temporal and spatial changes provide key data to reveal how these systems function at the biogeochemical level (e.g., Pakulski et al. 1995). There are abundant studies of nutrients in these systems (Seitzinger 1990), fewer studies of  $\text{O}_2$  (Smith and Mackiernan 1992), and even fewer examples where DIC and total alkalinity (TA) measurements have been made and related to nutrient dynamics (Hoppema 1990; Frankignoulle et al. 1996; Winter et al. 1996). Further, there is a significant lack of studies within low-salinity zones of estuaries, as noted by Hopkinson et al. (1999), particularly in systems that contain large areas of intertidal marshes. Investigators, often for logistical reasons, have concentrated either on what occurs in the intertidal or subtidal sediments or what occurs within the water column. Given the practical problems of the need for management of these systems and the lack of integrated scientific information with regard to how each component contributes to the flux of materials from the land to the sea, it is no wonder that after 40 yr of study, the argument still rages over the role of estuaries in modifying terrestrial/river inputs to the sea.

The estuaries and coastal waters of Georgia have been the subject of numerous ecological and biogeochemical studies since the 1950s (e.g., Ragotzkie and Pomeroy 1957; Teal 1962; Odum 1968; Pomeroy et al. 1972; Pomeroy and Wie-

bert 1981). Several investigations related to  $\text{O}_2$  and  $\text{CO}_2$  biogeochemistry have been conducted. For example, Hopkinson (1985), Griffith and Pomeroy (1995), and Pomeroy et al. (1993) consistently found that the total respiration rate was higher than that of total primary production in estuarine and coastal waters (i.e., the systems are heterotrophic). Geochemical studies of fluxes of organic carbon, nutrients, and major inorganic elements within Georgia rivers and the tidal freshwater zone were reported (e.g., Beck et al. 1974; Windom et al. 1975), and studies of nutrient and trace metal behavior in the estuaries and coastal shelf waters were conducted by Pomeroy et al. (1972) and Windom and Smith (1985), respectively.

There has been a major controversy over Odum's estuarine outwelling hypothesis regarding whether the marsh-surrounded estuary serves as a source of carbon and nutrients to coastal seas (Odum 1968; Chalmers et al. 1985; Hopkinson 1985; Hopkinson et al. 1989; Dame et al. 1986). However, these studies have been confined mostly to the near shore (Sapelo Island) marine system or to the high salinity portions of estuaries. In particular, the debate is largely based on studies conducted in the Duplin River, a blind tidal river adjacent to Sapelo Island, a system with no significant freshwater input.

Cai and Wang (1998) recently reported very high  $p\text{CO}_2$  values ( $>4,000 \mu\text{atm}$ ) in the low-salinity ( $<10$ ) zone of the Satilla River estuary in Georgia. These high  $p\text{CO}_2$  values imply that there is a large amount of  $\text{CO}_2$  loss to the atmosphere ( $70\text{--}100 \text{ mol m}^{-2} \text{ yr}^{-1}$ ), which is much higher, by at least a factor of 3, than can possibly be supported by the microbial respiration in the estuarine waters (about  $10\text{--}30 \text{ mol m}^{-2} \text{ yr}^{-1}$ , Cai and Wang 1998; Cai et al. 1999). Given the DIC concentration-salinity relationships, a large amount of DIC must be injected within the low-salinity zone. Cai and Wang (1998) suggested that the source of this DIC must be within the salt marshes. In this work, we provide new evidence (including data from tidal creeks) to support the

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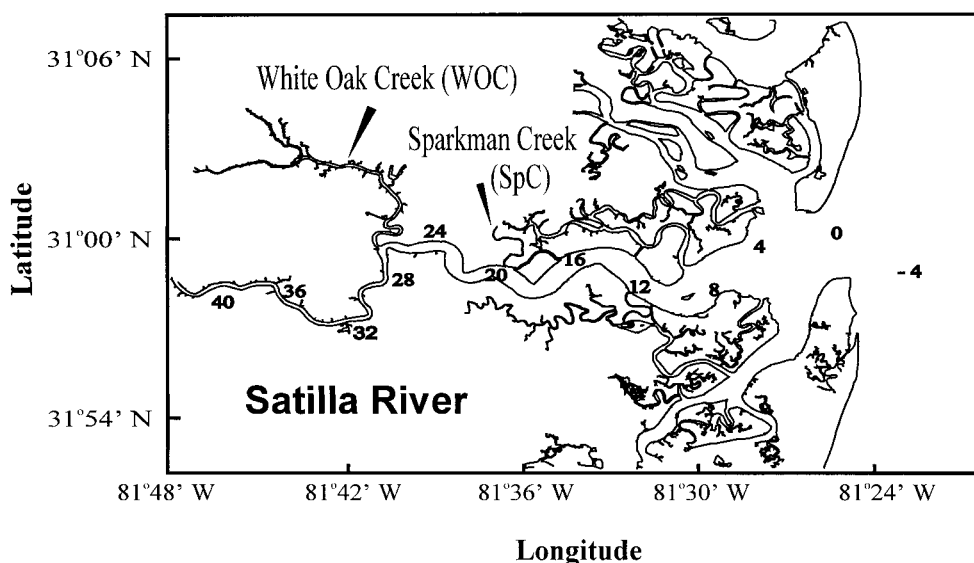


Fig. 1. Map of the Satilla River and the creeks. The estuary is about 40 km north of the boundary of Georgia and Florida. The station numbers are km from the river mouth.

hypothesis that the low-salinity zone of the intertidal marshes can be a major source of DIC for export to the coastal ocean and to the atmosphere via the estuary.

Historically, nutrient studies have played an important role in the biogeochemical studies of estuaries, particularly the dynamics of fixed nitrogen. Presently, nitrogen enrichment of estuaries and coastal oceans is considered a major problem for management agencies (Turner and Rabalais 1994). Nitrate and nitrite (hereafter designated  $\text{NO}_x$ ) concentrations are generally higher than ammonium ( $\text{NH}_4^+$ ) or phosphate in incoming river waters, and this is true of the Georgia rivers. In the Satilla River,  $\text{NO}_x$  enters the estuary at 2–10  $\mu\text{M}$ , rises to a peak value of 4–20  $\mu\text{M}$  at salinities around 10–15, and then decreases seaward (this paper and Wiebe and Sheldon unpubl. data). Generation of  $\text{CO}_2$  and  $\text{NO}_x$  is linked by the C/N ratio during organic matter decomposition. Therefore,  $\text{NO}_x$  may be exported from marshes together with DIC. Denitrification and anaerobic respiration within the marsh sediments, however, complicate the situation. The present study was conducted to define where  $\text{NO}_x$  is produced and the process by which it is consumed and to relate these processes to the  $\text{CO}_2$  fluxes.

In this paper, we examine the relationships between C and N in a high-humic (acidic), coastal plain estuary surrounded by extensive intertidal marshes. A box model is used to assist the discussion. These relationships help to delineate the complicated biogeochemical and hydrodynamic processes in such systems.

#### Sites and methods

Five major rivers, approximately evenly distributed along the coast of Georgia, intersect the coast, forming extensive saltmarsh estuaries and sounds protected by a series of barrier islands. The Satilla River, which originates within the coastal plain (Fig. 1), has a very high dissolved organic C content (25–30  $\text{mg L}^{-1}$ , mostly humic substances) and low

pH (<6) (Beck et al. 1974; Cai et al. 1998; Cai and Wang 1998). Extensive freshwater tidal marshes (up to 80 km from the river mouth) precede the estuarine salt marshes; zero salinity normally occurs at about 30–40 km from the river mouth. Estuarine mixing conditions are described in Alber and Sheldon (1999). River flow changes greatly seasonally and between years, with highest flows typically between January and April and lowest flows in the summer and fall (Fig. 2 and, for a 30-yr median, see Alber and Sheldon 1999). White Oak Creek (WOC), the site for our detailed creek studies, is the largest tidal creek of the river (Fig. 1) and is surrounded by extensive salt marshes. There is a clear salinity gradient within the creek.

The methods for collecting samples and determining pH, total alkalinity (TA), and DIC were described in Cai and Wang (1998). Briefly, pH was measured in a flow-through cell with an Orion Ross combination electrode within 5–30 min of sampling (precision 0.03 unit) at constant temperature near ambient. Uncertainties of pH measurement near zero salinity were discussed in Cai and Wang (1998). TA was determined by Gran titration, and DIC was determined by acid extraction and infrared detection on samples preserved by  $\text{HgCl}_2$ . Precision for both methods was 0.1–0.3%. Salinity was measured both by an Orion precision sensor and the shipboard conductivity, temperature, depth (CTD) probe. Agreement between the two methods was usually within 0.1 unit.

The conversion of nitrate to nitrite used the technique of Jones (1984), in which small nuggets of ‘spongy’ cadmium are shaken with the buffered sample. This technique was necessary because of the high humic acid content of the fresh and brackish waters of the Satilla River; twice the amount of cadmium as suggested by Jones (1984) was used to maintain acceptable reduction efficiency. Nitrite was determined using a standard Technicon autoanalyzer method (158–71W/B). Samples were routinely collected at slack low water.

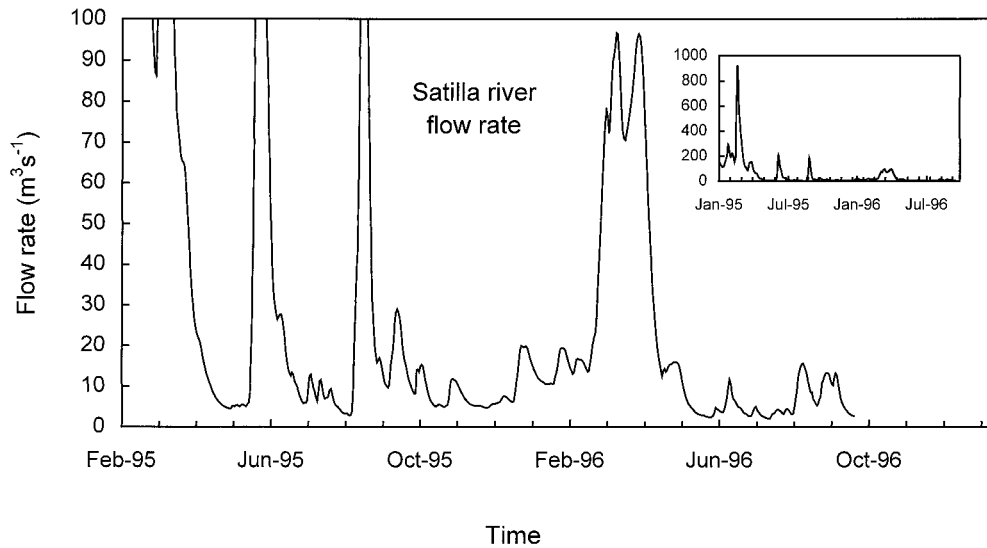


Fig. 2. Satilla River flow rate, January 1995–September 1996.

Results

In July 1996, pH increased with increasing salinity from a low river value (6.2) caused by humic substances to a high value (8.0) resulting from buffering by the seawater carbonate system. The river end-member pH value, however, was significantly (0.5–0.7 units) higher in July 1996 than in October 1995 (Cai and Wang 1998). DIC values in the low-

salinity zone were slightly higher than those of October 1995, whereas TA remained almost the same. The pH, DIC, and TA distributions (Fig. 3) in the estuary were generally similar to those in October 1995 (Cai and Wang 1998).  $\text{NO}_x$  increased sharply at low and medium salinities ( $S = 3\text{--}12$ ), then decreased to nearly zero at high salinity (Fig. 3D).

WOC had much higher values of DIC and TA and much lower concentrations of  $\text{NO}_x$  than waters of the same salin-

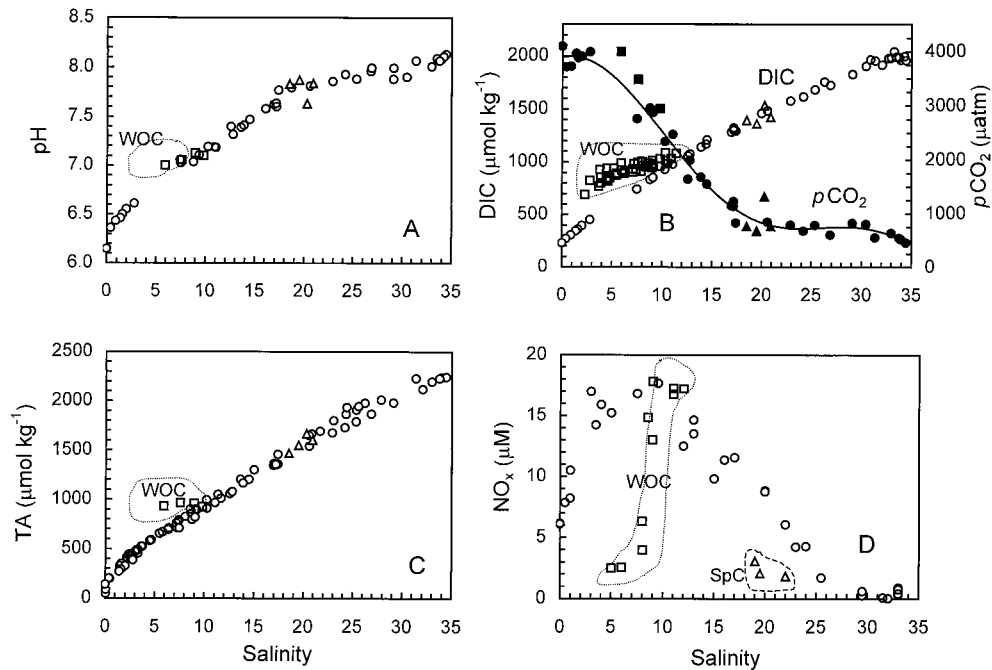


Fig. 3. Property-salinity graphs based on July 1996 data. Circles are the data in the main channel of the river, squares are from White Oak Creek (WOC), and triangles are from Sparkman Creek (SpC). In panel B, open symbols are DIC and solid symbols are  $p\text{CO}_2$ . All the data are original except  $p\text{CO}_2$ , which were reported before in Cai et al. (1999).  $p\text{CO}_2$  data were calculated from pH and DIC data reported here. The line ( $p\text{CO}_2$ ) is a six-order polynomial fitting of all original  $p\text{CO}_2$  data including those from WOC and SpC.

ities in the main channel (Fig. 3B, 3C, and 3D). However, pH values in WOC were similar to main channel waters at the same salinities (Fig. 3A). The  $p\text{CO}_2$  values from the same cruise (July 1996) were also similar for the creek and the river (Fig. 3B). This pattern may be interpreted as a control of pH by humic substances (Cai et al. 1998) and a  $\text{CO}_2$  loss to the atmosphere (Cai et al. 1999). There are no previous data from WOC for comparison.

Data from Sparkman Creek (SpC, represented by open or solid triangles at salinity  $\sim 20$ ) show a clear depletion in  $\text{NO}_x$  (Fig. 3D) but insignificant differences in pH, DIC, and TA values from those of the main channel (Fig. 3A, 3B, and 3C). SpC had a much higher salinity than WOC. At such high salinity, pH, DIC, and TA signals were already dominated by seawater, whereas the  $\text{NO}_x$  signal was still influenced by marsh processes.

A quick survey of the net removal or addition of  $\text{NO}_x$  and DIC in the estuary is instructive. In the standard estuarine mixing model (Boyle et al. 1974; Liss 1976; Li and Chan 1979; Officer 1979; Kaul and Froelich 1984), if the concentration ( $C$ ) of a dissolved constituent is continuous and at steady state over salinity coordinates, the net flux,  $F$ , of the dissolved constituent that passes any salinity ( $S$ ) isosurface is

$$F = R \left( C - S \frac{dC}{dS} \right) = RC^* \quad (1)$$

where  $R$  is the river flow rate. The term in brackets represents an effective river concentration ( $C^*$ ) of the constituent that would explain the concentration at that salinity if only conservative mixing between two end members were occurring.  $C^*$  may be obtained in a  $C$ - $S$  plot by extending the derivative at any salinity to zero salinity. The percentage removal at any point with respect to freshwater input is then defined as

$$\text{Removal (\%)} = \frac{C_0 - C^*}{C_0} \times 100 \quad (2)$$

Here  $C_0$  is the concentration at zero salinity.

For July 1996 data, there is a large addition of  $\text{NO}_x$  at  $S < 15$ . The net addition at  $S = 15$  calculated with Eq. 2 is as high as 350% with respect to the river input value (Fig. 4). Beyond salinity 15, there is a net in situ removal of  $\text{NO}_x$ , whereas the accumulated value with respect to river input is still an addition. At  $S = 30$ – $31$ , 100% of river-supplied  $\text{NO}_x$  and all  $\text{NO}_x$  produced between  $S = 0$ – $15$  have been removed (Fig. 4). The  $\text{NO}_x$  flux to coastal seas from the Satilla River is nearly zero because the effective  $\text{NO}_x$  concentration is nearly 0 or slightly negative around salinity 33 (Fig. 4).

For DIC data (excluding data from WOC, Fig. 3B), a simple linear regression between salinities 15 and 30 results in an effective DIC concentration of about  $580 \mu\text{M}$  at  $S = 0$ , which is similar to October 1995 (Cai and Wang 1998). The amount of DIC exported ( $3$ – $3.5 \times 10^8 \text{ mmol C d}^{-1}$ ) from the estuarine system to the coastal sea is a product of this effective concentration and the river flow rate ( $6$ – $7 \text{ m}^3 \text{ s}^{-1}$ ). The higher effective DIC concentration ( $580 \mu\text{M}$  or  $\text{mmol m}^{-3}$ ), as opposed to the actual river DIC concentration ( $200 \mu\text{M}$ ) at  $S = 0$ , reflects the net result of DIC addition from intertidal marshes. Most of the total DIC addition, however,

is lost to the atmosphere (estimated as  $\sim 3.3 \times 10^9 \text{ mmol C d}^{-1}$  in Table 1) and only about 10% is exported to the coastal sea. The above data analysis, though instructive, does not specify sources and sinks from a biogeochemical perspective and provides no internal link between C and N. Therefore, a box model is used to assist further discussion. The principle of a box model is the same as the continuous model, both based on the mass balances of a conservative salt and nonconservative chemicals. It is, however, easier to specify sources/sinks and rates of biogeochemical productions/consumptions in a box model.

### The box model

We modeled the  $\text{NO}_x$  concentration distribution curve based on known biogeochemical mechanisms and fluxes and hydrodynamics, since by comparison with field data, such an approach allows evaluation and improvement of our knowledge of the biogeochemical processes functioning in the system. Officer (1980) has presented a useful modeling tool, the box model, which only requires inputs of salinity distribution, river flow rate, and biogeochemical reaction terms or fluxes.

For a  $k$ -box estuary (Fig. 5) that has a mass or flux gain of  $W_x$  ( $\text{mmol d}^{-1} \text{ box}^{-1}$ ) of a chemical in box  $x$  ( $x = 1$  to  $k$ ), the total concentration deviation ( $\Delta C_y$ ) in box  $y$  from a conservative (linear) mixing of the two end members is the sum of the contributions from all the boxes (1 to  $k$ ):

$$\begin{aligned} \Delta C_y = & \frac{S_{\text{sw}} - S_y}{S_{\text{sw}} - S_0} \\ & \times \left[ W_1 \frac{S_1 - S_0}{S_1} + \dots + W_{y-1} \frac{S_{y-1} - S_0}{S_{y-1}} + 0.5 W_y \frac{S_y - S_0}{S_y} \right] \frac{1}{R} \\ & + \frac{S_y - S_0}{S_{\text{sw}} - S_0} \left[ 0.5 W_y \frac{S_{\text{sw}} - S_y}{S_y} \right. \\ & \left. + W_{y+1} \frac{S_{\text{sw}} - S_{y+1}}{S_{y+1}} + \dots + W_k \frac{S_{\text{sw}} - S_k}{S_k} \right] \frac{1}{R} \end{aligned} \quad (3)$$

where sw and 0 indicate the sea and the river end members, respectively. The first half of Eq. 3 summarizes the contributions of all the  $x$  boxes upstream of  $y$  (including half of  $y$ ), and the second half summarizes the contributions of all the  $x$  boxes downstream of  $y$  (including half of  $y$ ). As illustrated in Fig. 5, since only the nonconservative deviation of the concentration is being considered, zero concentrations are assigned for the two end members. To help understand the model, a simple geometric derivation and explanation of Eq. 3 is given here. A detailed algebraic derivation was given in Officer (1980).

For a source box,  $x$ , with a constant flux gain of  $W_x$ , the concentration change in box  $x$  is

$$\Delta C_x = \frac{W_x \tau_x}{V_x}, \quad (4)$$

where  $\tau_x$  is the residence time of water in box  $x$ . The volume

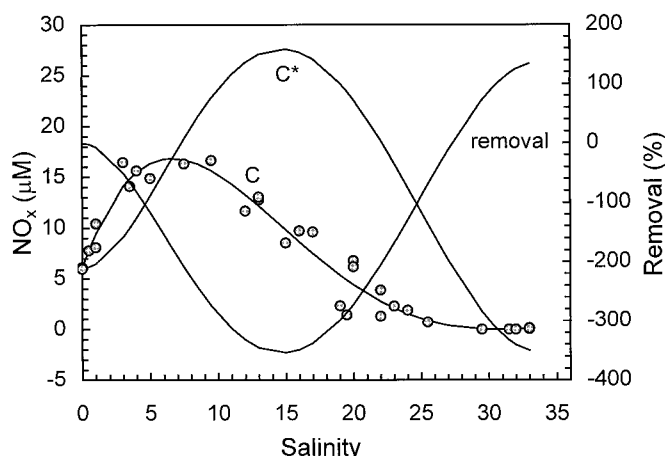


Fig. 4. Effective concentration ( $C^*$ ) and removal (percentage) of  $\text{NO}_x$  in the Satilla estuary based on June 1996 data. The polynomial fitting equation for all  $\text{NO}_x$  data is  $C = 0.0000012825 \times S^5 - 0.00024785 \times S^4 + 0.016071 \times S^3 - 0.42779 \times S^2 + 3.8203 \times S + 6.0815$ .

( $V_x$ ) of box  $x$  can be related to the river water volume ( $V_R$ ) in  $x$  and the salinities (assume  $S_0 = 0$  for now), as  $S_x = S_{sw}V_{sw}/(V_R + V_{sw})$ . Rearranging this equation, we have  $V_{sw} = V_R S_x / (S_{sw} - S_x)$ . Therefore, we have  $V_x = V_R + V_{sw} = V_R S_{sw} / (S_{sw} - S_x)$ . Inserting  $V_R = R\tau_x$ , we have  $V_x = R\tau_x S_{sw} / (S_{sw} - S_x)$ . Substituting this result into Eq. 4, we have

$$\Delta C_x = \frac{W_x S_{sw} - S_x}{R S_{sw}} \quad (5)$$

When the salinity at the river end,  $S_0$ , does not equal zero, Eq. 5 is modified as

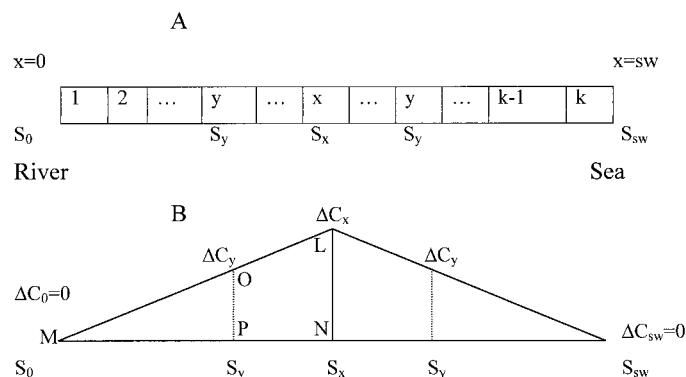


Fig. 5. (A) Illustration of a  $k$ -box model of an estuary. (B) Geometric illustration of the mixing of a chemical signal showing the linear dependence of  $\Delta C_y$  on  $\Delta C_x$  ( $y$  is either upstream or downstream of  $x$ ). Note the two triangles ( $\triangle LMN$  and  $\triangle OMP$ ) are similar, and the signal decreases to 0 at the two boundaries.

$$\Delta C_x = \frac{W_x S_x - S_0 S_{sw} - S_x}{R S_x S_{sw} - S_0} \quad (\text{Officer 1980}) \quad (6)$$

The concentration change that a source box  $x$  brings to box  $y$  is only a linear dilution of  $\Delta C_x$ . By geometric principle, as shown in Fig. 5B, triangle  $LMN$  and triangle  $OMP$  are similar, and therefore  $\Delta C_y / \Delta C_x = OP / LN = MP / MN = (S_y - S_0) / (S_x - S_0)$ . If box  $y$  is upstream of box  $x$  (left in Fig. 5B), we have

$$\Delta C_y = \Delta C_x \frac{S_y - S_0}{S_x - S_0} = \frac{W_x S_{sw} - S_x S_y - S_0}{R S_x S_{sw} - S_0}, \quad (7)$$

if box  $y$  is downstream of box  $x$  (right in Fig. 5B), we have

Table 1. Various properties of the 5-km sections of river and marsh (or boxes) (July 1996 case). The left panel lists all the basic data for each estuarine section. The marsh  $\text{NO}_x$  generation rate was calculated by multiplying the nitrification rate ( $11.0 \text{ mmol m}^{-2} \text{ d}^{-1}$ ) by marsh areas and a factor of 1/4, since marshes are inundated only 6 h a day. This is consistent with the calculation of  $\text{CO}_2$  mass balance (Cai et al. 1999). Since all  $\text{NO}_x$  was removed (*see text*), the total  $\text{NO}_x$  generation rate ( $5.2 \times 10^8 \text{ mmol d}^{-1}$ ) is also the total removal rate in the marsh. In the right panel, box  $\text{NO}_x$  flux =  $16/106 \times \text{CO}_2$  flux. Box  $\text{CO}_2$  fluxes were calculated according to Cai et al. (1999)[i.e., box flux = unit area flux  $\times$  (river area +  $1/4 \times 1/6 \times$  marsh area), where the factor 1/6 reflects that marshes have a much reduced wind speed and marsh waters are stagnant.]

Distance (m)	Box	Salinity	River area $10^6 \text{ m}^2$	Marsh area $10^6 \text{ m}^2$	Depth (m)	Marsh $\text{NO}_x$ generation $10^6 \text{ mmol d}^{-1}$	$\text{CO}_2$ flux $\text{mmol m}^{-2} \text{ d}^{-1}$	Box $\text{CO}_2$ flux $10^6$ $\text{mmol d}^{-1}$	Box $\text{NO}_x$ generation $10^6 \text{ mmol d}^{-1}$	Model $W_x 10^6$ $\text{mmol d}^{-1}$
Sea		34.6								
-5-0	10	34.00	18.7	0.00E	4	0	10	187	28.3	-2.00
0-5	9	32.80	17.0	15.0	4	41.3	10	177	26.7	-3.00
5-10	8	30.50	9.07	22.0	3.6	60.5	25	250	37.7	-2.50
10-15	7	24.00	5.83	23.0	3.3	63.3	25	170	25.6	-2.00
15-20	6	17.50	4.66	25.0	2.9	68.8	40	228	34.4	-1.00
20-25	5	13.00	3.26	21.0	3.5	57.8	100	414	62.5	1.00
25-30	4	7.00	3.06	46.0	6	12.7	170	846	12.8	3.50
30-35	3	5.00	1.05	11.0	6.5	30.3	230	346	52.3	3.50
35-40	2	2.00	1.10	14.0	6	38.5	230	386	58.3	1.50
40-45	1	0.30	1.00	12.0	6	33.0	220	330	49.8	1.00
River	0	0.1								
Box total		—	64.8	189	—	520	—	3330	503	0

$$\Delta C_y = \Delta C_x \frac{S_{sw} - S_y}{S_{sw} - S_x} = \frac{W_x S_x - S_0 S_{sw} - S_y}{R S_x S_{sw} - S_0} \quad (8)$$

To obtain the total concentration change in box  $y$  caused by removal/addition in all boxes, we need to add all of source box  $x$  contributions upstream of box  $y$  (Eq. 7) and downstream of  $y$  (Eq. 8). The result is Eq. 3 (note only  $y$  is used for the box notation in Eq. 3).

The calculation of  $\Delta C_y$  thus becomes a simple operation in spreadsheet software if the flux to each box,  $W_x$ , is known. In such cases, the nonconservative mixing of a chemical element in an estuary is actually treated as a summation of conservative mixing between any two boxes as expressed in Eq. 3 and as illustrated in the geometric explanation (Fig. 5B). This is because any addition or removal during the mixing is already reflected in the flux terms of the other boxes. If the flux terms are not known independently, they may be expressed as first-order reactions with respect to the concentration. Then an iterative method is needed to solve a set of  $k$  equations simultaneously (Officer 1980). It is, however, important to understand that  $\Delta C_y$  represents only the nonconservative part of estuarine mixing. In other words, it is the concentration deviation of a nonconservative constituent in box  $y$  from a conservative mixing behavior (i.e., linear mixing). Therefore, in comparison with field data (i.e.,  $C_y$ ), a linear mixing term from the two end members should be added to  $\Delta C_y$ .

We constructed a 10-box model for the Satilla River estuary-intertidal marsh system and analyzed field results from July 1996, October 1995, and April 1995. The length of each box is 5 km, from  $-5-0$  km (box 10) to  $40-45$  km (box 1). Since this is a one-dimensional model, water in a section of the river and its surrounding marshes is considered as one box. This specification is not ideal but is adequate for illustrating sufficient details of the key biogeochemical features of the system.

The model requires salinity data, and Fig. 6 shows salinity distributions measured in July 1996 and April and October 1995. There was a large salinity difference between high water and low water at any given fixed station. We used low water salinities (for July 1996 case *see Table 1*) read from Fig. 6, since most of our C and N data were measured at low water. The geochemical application of the model will be given in the next section.

## Discussion

*Intertidal marshes as a source of DIC and a sink of  $NO_x$* —Based on the DIC distribution in the Satilla River, in particular its relationship to White Oak Creek (WOC), Cai and Wang (1998) speculated that a large amount of DIC must be exported from salt marshes via WOC. DIC concentrations in the creek waters reported here were up to twice as high as those in the Satilla River waters at similar salinities (Fig. 3B). Clearly, this salt marsh represents a significant source of inorganic carbon to the estuary. If one looks only at the concentration-salinity plot of  $NO_x$  (Fig. 3D) in the river, it appears that at the same position of DIC and TA increase (output from WOC),  $NO_x$  also should be exported from the creek. However, this was clearly not the case when we ex-

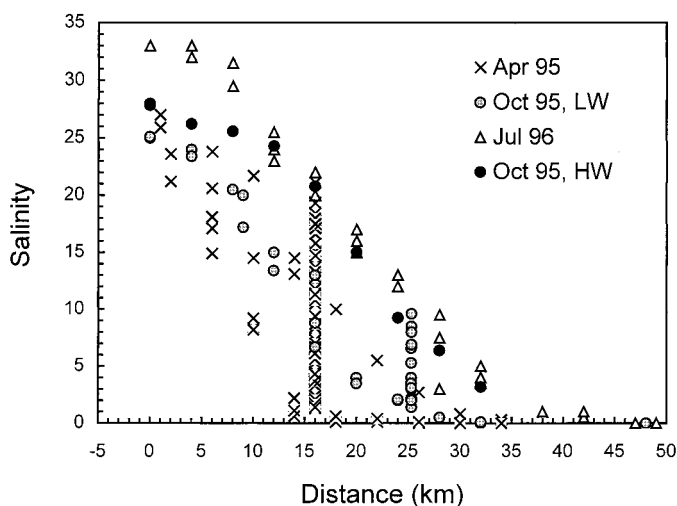


Fig. 6. Salinity distributions during the three cruises that  $NO_x$  data were collected. Some anchor station data (including both high and low water) at 5, 15, and 25 km are included to show the range of variations. For October 1995, high and low water data are presented separately with the 25 km anchor station data plotted as low water. Only low water (LW) data were used in the model.

tended our sample coverage into WOC and SpC:  $NO_x$  was depleted to nearly zero in both creeks (Fig. 3D).

We also exclude  $NH_4^+$  export from WOC as a significant cause for the  $NO_x$  peak in the estuary. The high respiration rate and low oxygen content in the marsh sediments suggest that  $NH_4^+$  is probably the primary N form released from the sediments. However, since the marsh waters and certainly the creek waters are always oxic, we believe all the  $NH_4^+$  is eventually used or nitrified and subsequently denitrified by microorganisms within the marsh and tidal creeks. This is suggested from the WOC data, since  $NH_4^+$  concentrations were only  $0-1.2 \mu M$  in creek waters in July 1996 and were lower than that of the main channel water (Wiebe and Sheldon unpubl. data). Ammonium release from marshes adjacent to the riverbank was not measured and potentially could contribute to the  $NO_x$  accumulation in the river water. However, based on many flume experiments in the southeastern U.S., Childers (1994) and Childers et al. (1993) concluded that marshes in this area take up  $NH_4^+$  and  $NO_x$ . Therefore, the  $NO_x$  peak in the low-salinity part of the main channel water is most likely produced in situ by water column metabolism. Another potential source of  $NO_x$  production could be the oxidation of  $NH_4^+$  released from the riverbed sediments.

Several other lines of evidence suggest that the DIC increase and  $NO_x$  removal are largely due to processes in marsh sediments. First, higher TA appears concurrently with higher DIC in creek water (Fig. 3C), which suggests significant anaerobic degradation ( $SO_4$  reduction) of organic matter (Bernier 1982). Since the  $O_2$  content of marsh waters was never near 0 (except in isolated areas, Cai et al. 1999), significant anaerobic metabolism could only occur in marsh or subtidal sediments. Second, benthic chamber fluxes of DIC and  $O_2$  in marsh sediment measured at Sapelo Island, Georgia, and from the low-salinity marsh of WOC were higher

than depth-integrated estuarine water respiration (Cai et al. 1999). Third,  $\text{NO}_x$  concentrations in WOC and SpC were much lower than those in the main channel at similar salinities, which indicates intense removal within the marshes. Although assimilation and denitrification are both possible, we believe that denitrification is the most reasonable hypothesis for the  $\text{NO}_x$  removal in the marshes. Since marsh waters are oxygenated, significant denitrification most likely occurs in the sediments.

*Rates of organic carbon respiration and the N cycle in marshes*—Two approaches are taken to estimate nitrification and denitrification rates in the marsh and the river. The first approach is based on field-determined  $\text{CO}_2$  generation rates within the intertidal marshes, whereas the second approach is based on atmospheric  $\text{CO}_2$  fluxes (net system respiration rates) within the entire estuary. First the predicted  $\text{NO}_x$  generation in the marshes is calculated from  $\text{CO}_2$  generation rates. The C/N ratio during aerobic respiration is 106/16 (assuming a Redfield ratio as a low boundary; higher C/N ratios will be discussed later). Given a  $\text{CO}_2$  generation rate of  $42 \text{ mmol m}^{-2} \text{ d}^{-1}$  from sediment (a mean value used in the  $\text{CO}_2$  and  $\text{O}_2$  mass balance model of Cai et al. 1999) and  $61 \text{ mmol m}^{-3} \text{ d}^{-1}$  in marsh waters (assuming a water depth of 0.5 m), the  $\text{NO}_x$  generation rate in marshes would have to be  $11.0 \text{ mmol m}^{-2} \text{ d}^{-1}$  ( $[42 + 61 \times 0.5] \times 16/106$ ). Since the  $\text{NO}_x$  concentration was near zero in marsh waters, the denitrification rate must be equal or higher than this nitrification rate to remove the  $\text{NO}_x$  produced in the marsh and imported from the river via tidal action. The estimated nitrification and denitrification rates in the marsh should be viewed as the upper limit. The C/N ratio of macrophytes can be much higher than the Redfield ratio, for example, Haines et al. (1977) reported a C/N ratio of 32–54 in *Spartina* plants and a ratio of 10–15 in the sediments at Sapelo Island. Hopkinson et al. (1999) reported a C/N ratio between 11 and 23 in sediments of the Parker River, Massachusetts. However, the estimated C/N ratio (6.8–10.2) in the respiration products in Hopkinson et al. (1999) is not drastically different from the Redfield ratio (6.6). If we use a C/N ratio higher than the Redfield ratio, the nitrification rate and therefore the required denitrification rate would be reduced accordingly. If we assign the high end of the C/N = 23 in sediments as the C/N ratio in the respiration products, the lowest denitrification rate one could expect should be  $3.15 \text{ mmol m}^{-2} \text{ d}^{-1}$ .

There are no direct measurements of the denitrification rate for the Satilla River marsh sediments. However, in nearby Sapelo Island marshes, the denitrification rate averaged  $12.7 \text{ mmol m}^{-2} \text{ d}^{-1}$  (Whitney et al. 1981). Thus the estimated denitrification rate of 3–11  $\text{mmol m}^{-2} \text{ d}^{-1}$  for the Satilla marshes based on the  $\text{CO}_2$  generation rate appears to be a reasonable value.

*Constraints based on estuarine processes*—The second approach to constrain nitrification and denitrification rates is based on the comparison of  $\text{NO}_x$  concentrations measured in the main estuarine channel with those predicted from water-to-air  $\text{CO}_2$  fluxes using a simple mass balance calculation or the box model. If we assume that the estuarine-intertidal marsh complex is near steady-state, then all of the  $\text{CO}_2$  lost

to the air would have to be supplied by internal processes. Cai et al. (1999) demonstrate that most of the lost  $\text{CO}_2$  (80%) is exported from the marshes to the estuary via tidal action. Other processes that contribute to  $\text{CO}_2$  flux include photosynthesis and respiration in the water column and respiration in the riverbed. We further assume  $\text{CO}_2$  flux to each section of the estuary is balanced approximately by  $\text{CO}_2$  flux to the air (Cai et al. 1999). This is only an approximation. Owing to tidal mixing,  $\text{CO}_2$  input into a certain section of the estuary does not necessarily equal the water-to-air flux locally. In addition, there is also a small portion of DIC that is exported to the coastal sea.

$\text{CO}_2$  flux in each 5 km of river and marsh (box) was calculated according to Cai et al. (1999) and a total water-to-air flux of  $3.3 \times 10^9 \text{ mmol d}^{-1}$  was obtained for the entire estuary (Table 1). This value translates into an  $\text{NO}_x$  generation rate of  $5.0 \times 10^8 \text{ mmol d}^{-1}$  with a C/N = 6.6 (or  $1.4 \times 10^8 \text{ mmol d}^{-1}$  with a C/N = 23). With a residence time of longer than 60 d (Alber and Sheldon 1999) and a total estuarine volume of  $2.58 \times 10^8 \text{ m}^3$ , this amount of  $\text{NO}_x$  should generate, at least, an average  $[\text{NO}_x] = 116 \mu\text{M}$  ( $= 5.0 \times 10^8 \text{ mmol d}^{-1} \times 60 \text{ d} / 2.58 \times 10^8 \text{ m}^3$ , or  $33 \mu\text{M}$  using C/N = 23) if the produced  $\text{NO}_x$  is not removed. The average  $[\text{NO}_x]$  measured in the Satilla during July 1996 was only  $3.0 \mu\text{M}$  ( $= \sum C_i V_i / \sum V_i$ , where  $C_i$  is taken from Fig. 3D and  $V_i$  is the volume of a section of estuary [box  $i$ ] calculated from the depth and area given in Table 1). Therefore, this rough calculation shows that most of the produced  $\text{NO}_x$  must be removed within the estuarine system.

The total nitrification rate in the marshes estimated in the previous section (rate  $\times$  total marsh area =  $5.2 \times 10^8 \text{ mmol d}^{-1}$ ) is close to the estimated  $\text{NO}_x$  generation from the water-to-air  $\text{CO}_2$  flux in the entire estuarine system ( $5.0 \times 10^8 \text{ mmol d}^{-1}$ ). We do not expect these numbers to be matched perfectly, since each estimate has a large uncertainty depending on several factors. Further, we expect that the uncertainty in estimating the net  $\text{NO}_x$  generation flux into a box ( $W_x$ ) by subtracting denitrification from the total  $\text{NO}_x$  generation in the box will be very large, since the two numbers are the same order of magnitude. Instead, we take an inverse approach. By matching the model output to the measured  $\text{NO}_x$  concentration (Fig. 7), we can constrain  $W_x$  (last column of Table 1). These fluxes are on the order of  $1 \times 10^6 \text{ mmol d}^{-1}$  and are 10 to 50 times smaller than the  $\text{NO}_x$  generation calculated from the  $\text{CO}_2$  fluxes in the boxes. The summation of these fluxes ( $\sum W_x$ ) is approximately zero with addition in the upstream and removal in the downstream, which is consistent with the simple survey of the  $\text{NO}_x$  distribution using Eq. 2. The conclusion is again that all  $\text{NO}_x$  generated during organic matter degradation in the estuarine-marsh complex is removed inside the system.

Since  $\text{NO}_x$  production rates estimated from marsh processes only and from gas fluxes in the entire system are roughly the same, we propose that  $\text{NO}_x$  production and removal occurs mostly within the marshes. As discussed earlier, the  $\text{NO}_x$  peak in the low-salinity part of the main channel water is most likely produced in situ by water column metabolism and, probably, oxidation of  $\text{NH}_4^+$  released from riverbed. A significant part of this river-generated  $\text{NO}_x$  is

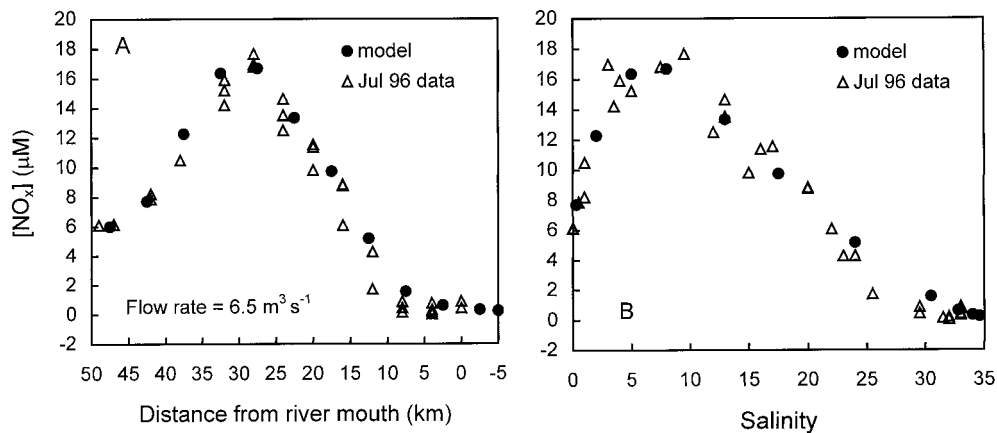


Fig. 7. Model and field data of  $\text{NO}_x$ -distance/salinity distributions in July 1996.

carried into marshes by tidal action and is denitrified in marsh sediments.

*Variations of  $\text{NO}_x$  concentrations and fluxes*—Concentrations of  $\text{NO}_x$  in October 1995 and April 1995 (Fig. 8) are much lower than in July 1996, and the peaks of  $\text{NO}_x$  concentrations are shifted downstream. Biogeochemical parameters should vary seasonally and therefore should contribute to the observed differences. For example, based on temperature alone we might expect a smaller respiration rate in spring than in summer. However, since we argued that all or most of the  $\text{NO}_x$  production was consumed by denitrification

in the marsh and the supply of extra  $\text{NO}_x$  to the marsh was controlled by the tidal action and freshwater input, the seasonal variation of net  $\text{NO}_x$  consumption by marshes should depend much less on temperature or light but rather on river flow rates and tidal actions.

Variations in the river flow rate are critically important in affecting the  $\text{NO}_x$  distribution. As shown in Fig. 2, the river flow rate varied greatly over the year. July 1996 was a particularly dry period, with a very low river flow rate of 6–7  $\text{m}^3 \text{s}^{-1}$ , whereas October 1995 had a higher river flow rate of about 15  $\text{m}^3 \text{s}^{-1}$ . During April 1995, the river flow rate reached 50  $\text{m}^3 \text{s}^{-1}$ . Using a current flow rate during the pe-

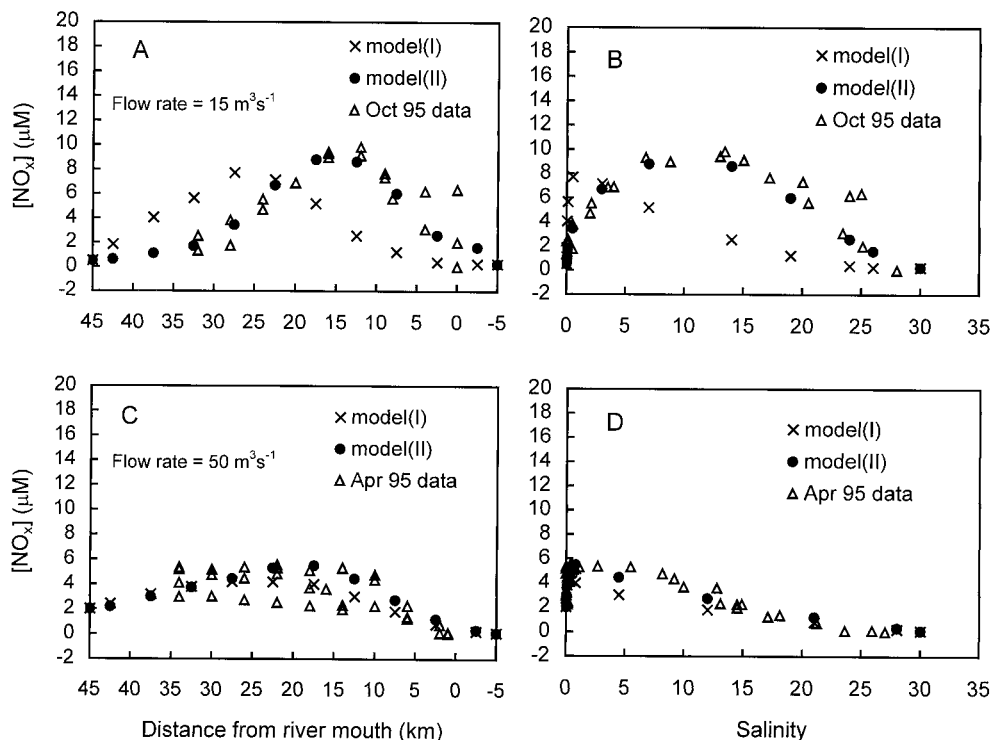


Fig. 8. Model and field data of  $\text{NO}_x$ -distance/salinity distributions in October and April 1995. Model (I) uses the same fluxes as in July 1996 (Fig. 7) for April and October 1995, whereas model (II) produces the best fit of the data.

riod of data collection is overly simplified since the Satilla estuary has a long residence time (or flushing times, *see* Alber and Sheldon 1999 for a discussion of this issue). The residence time was estimated as long as over 4 months during low flow periods to as short as less than 1 month during high flow periods (Alber and Sheldon 1999). Figure 8 presents field  $\text{NO}_x$ - $S$  relationships together with model simulations for April 1995 and October 1995. As expected,  $\text{NO}_x$  concentrations in the estuarine water decrease as flow rate increases. The  $\text{NO}_x$  concentration distribution is more sensitive to flow rate variation during the low flow case. When the flow rate is above  $50 \text{ m}^3 \text{ s}^{-1}$ , distribution of  $\text{NO}_x$  concentration in the estuarine waters of the Satilla is no longer sensitive to flow rate variation.

Another notable feature is that  $\text{NO}_x$  production is not significant in the freshwater zone. This is evident from much higher model-generated values (model [I]) than the field data in the freshwater zone for October 1995 when the same set of fluxes ( $W_x$ ) used in July 1996 were used for the model simulation. Owing to a high flow rate, salinities upstream of 27 km were nearly zero. Zero  $\text{NO}_x$  production rates must be assigned to these boxes in order to generate a simulation curve (model [II]) that best fits the data. For the April 1995 case, the differences between model (I) and the data are not very significant since the very high flow rate has suppressed the other signals.

As mentioned earlier, the box model simulation shows that net  $\text{NO}_x$  export to the coastal sea is nearly zero for July 1996. Best fit of field data (model [III]) generates a total net seaward flux of  $13 \pm 1 \times 10^6 \text{ mmol d}^{-1}$  for October 1995 and  $6 \pm 1 \times 10^6 \text{ mmol d}^{-1}$  for April 1995. The total estuarine  $\text{NO}_x$  production rates in these seasons are not known, since  $\text{CO}_2$  fluxes were not determined. If a range of  $100\text{--}500 \times 10^6 \text{ mmol d}^{-1}$  (20–100% of July 1996 value) is assumed for  $\text{NO}_x$  production rate in all seasons, then even during October 1995 the export flux is only a few percent of total  $\text{NO}_x$  production rate in the entire estuary. Thus, in all three cruises, the role the marshes play in the removal of internally produced  $\text{NO}_x$  is similar, whereas the river flow rate variations clearly control the seasonal differences in the  $\text{NO}_x$  distribution in the Satilla River.

The seaward  $\text{NO}_x$  flux in October 1995 is, however, significant when compared to freshwater  $\text{NO}_x$  input. The total freshwater  $\text{NO}_x$  fluxes (flow rate times river concentration) are 3.4, 1.9, and  $13 \times 10^6 \text{ mmol d}^{-1}$ , whereas export fluxes constrained from the box model are 0, 13, and  $6.5 \times 10^6 \text{ mmol d}^{-1}$ , respectively, for July 1996, October 1995, and April 1995. Therefore, the  $\text{NO}_x$  exports with respect to riverine fluxes are 0%, 500%, and 50%, respectively. This last conclusion may be checked quickly with a continuous model (Eq. 2) by comparing  $C^*$  with  $C_0$ . Table 2 lists river concentrations, effective concentrations, and fluxes at the river mouth generated from the continuous model (Eqs. 1 and 2). The two approaches agree in general. However, in the continuous model, estimated export rate at the river mouth (0 km) is very sensitive to a few data points measured in the high salinity zone. This is particularly true for October 1995 and April 1995, when concentration gradients near the river mouth were far from zero. For July 1996, such estimation has much less uncertainty since the gradient is nearly zero

Table 2. The effective concentration and seaward fluxes of  $\text{NO}_x$  at the river mouth during three field campaigns. Current flow rates during the campaigns are used without consideration of flow rate before the campaign. A 10–20% uncertainty applies to freshwater fluxes, and a 50% uncertainty applies to the effective fluxes. Also given is percentage of freshwater  $\text{NO}_x$  exported from the river mouth.

	Campaign		
	July 1996	October 1995	April 1995
Flow rate ( $\text{m}^3 \text{ s}^{-1}$ )	6.5	15	50
River concentration ( $\mu\text{M}$ )	6	1.5	3
Freshwater flux ( $10^6 \text{ mmol d}^{-1}$ )	3.4	1.9	13
Effective concentration ( $\mu\text{M}$ )	–2–0	~10	~1.5
Effective flux ( $10^6 \text{ mmol d}^{-1}$ )	–1–0	13	6.5
Exported (%)	–33–0	567	50

(Fig. 4). There is no such dependence on end-member values in the box model, since fluxes are constrained for each individual box.

In summary, our water column data from the tidal creeks and the main channel have shown that intertidal marshes are a significant source of DIC and TA and a sink for  $\text{NO}_x$ .  $\text{NO}_x$  was produced in low-salinity waters of the estuary and subsequently removed in the marshes during tidal excursion. Nearly all of the  $\text{NO}_x$  entering the estuary from the Satilla River and generated within the estuary (including the marsh) was removed by the time the estuarine water reached the sea. We reached this conclusion by coupling C and N mass balance calculations through the organic matter C/N ratio using two approaches. First, in the marsh, a significant DIC generation rate ( $72 \text{ mmol m}^{-2} \text{ d}^{-1}$ ) and a complete removal of  $\text{NO}_x$  lead to a removal (most likely via denitrification) rate of  $3\text{--}11 \text{ mmol m}^{-2} \text{ d}^{-1}$  with a C/N ratio between 23 and 6.6. Second, modeling  $[\text{NO}_x]$  in estuarine waters from the  $\text{CO}_2$  fluxes from water to air in various sections of the estuary also requires a complete removal of  $\text{NO}_x$  within the estuary.

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