



## Tidal dynamics and rainfall control N<sub>2</sub>O and CH<sub>4</sub> emissions from a pristine mangrove creek

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[1] Dissolved CH<sub>4</sub>, N<sub>2</sub>O, O<sub>2</sub>, and inorganic nitrogen nutrients (NH<sub>4</sub><sup>+</sup>, NO<sub>3</sub><sup>-</sup> and NO<sub>2</sub><sup>-</sup>) were measured over tidal cycles in pristine Wright Myo mangrove creek waters during dry and wet seasons. Dissolved CH<sub>4</sub> and N<sub>2</sub>O showed no seasonality (dry season; 491 ± 133 nmol CH<sub>4</sub> l<sup>-1</sup>, 9.0 ± 2.3 nmol N<sub>2</sub>O l<sup>-1</sup>, wet season; 466 ± 94 nmol CH<sub>4</sub> l<sup>-1</sup>, 8.6 ± 1.3 nmol N<sub>2</sub>O l<sup>-1</sup>). Creek water dissolved gas and inorganic nitrogen distributions reflect sediment porewater release during hydrostatic pressure drop toward low water. Creek water CH<sub>4</sub> emission was suppressed by oxidation during rainfall, consistent with changes to dissolved nitrogen speciation, although N<sub>2</sub>O emissions were unaffected. Scaling up emissions flux estimates from mangrove creek waters and intertidal sediment gives worldwide mangrove emissions ~1.3 × 10<sup>11</sup> mol CH<sub>4</sub> yr<sup>-1</sup> and 2.7 × 10<sup>9</sup> mol N<sub>2</sub>O yr<sup>-1</sup>; mangrove ecosystems are thus small contributors to coastal N<sub>2</sub>O emissions but could dominate coastal CH<sub>4</sub> emissions. Comparing our data with mangrove CO<sub>2</sub> fluxes, mangrove ecosystems could be small net contributors of atmospheric greenhouse gases. **Citation:** Barnes, J., R. Ramesh, R. Purvaja, A. Nirmal Rajkumar, B. Senthil Kumar, K. Krithika, K. Ravichandran, G. Uher, and R. Upstill-Goddard (2006), Tidal dynamics and rainfall control N<sub>2</sub>O and CH<sub>4</sub> emissions from a pristine mangrove creek, *Geophys. Res. Lett.*, 33, L15405, doi:10.1029/2006GL026829.

### 1. Introduction

[2] Mangroves cover ~75% of tropical coastlines and are among the world's most productive ecosystems. About one half of mangrove net primary production is exported to coastal shelves [Dittmar *et al.*, 2006]; the remainder is buried and/or recycled, some being exported to the atmosphere from heterotrophic mangrove waters and sediment, as CO<sub>2</sub>, N<sub>2</sub>O and CH<sub>4</sub>. Measurements of these emissions are relatively few and no simultaneous studies of fluxes from both mangrove forest sediment and surrounding creek waters exist. We discuss CH<sub>4</sub>, N<sub>2</sub>O, O<sub>2</sub> and inorganic nitrogen dynamics, and CH<sub>4</sub> and N<sub>2</sub>O emissions fluxes from pristine mangrove forest sediment and creek waters estimated during dry and wet season tidal cycles. Combining these with available emissions fluxes of CO<sub>2</sub>, we

discuss the potential contribution of mangrove ecosystems to the greenhouse gas budget of the atmosphere.

### 2. Sites and Methods

[3] Wright Myo (11°47'27.7"N, 92°42'24.3"E) is a small, tropical tidal mangrove creek (~8 km long, 20–40 m wide dependant on tidal state) on South Andaman Island, 1200 km east of mainland India. The Andaman Island mangroves are sparsely populated and among the worlds most pristine. Vegetation is dominated by dense *Rhizophora* sp. stands naturally shielded from strong winds and waves; the surrounding area is mainly hilly, forest vegetation.

[4] We sampled over full tidal cycles: 12:30–11:30 hrs, 23–24 January 2004 (dry season); 13:00–12:00 hrs, 21–22 July 2004 (wet season). Mean tidal ranges and water depths respectively were: January, 0.07–2.21 m and 3.2 m; July 0.69–2.77 m and 3.4 m. For both surveys high tide was around midnight, with low tides around sunset and sunrise.

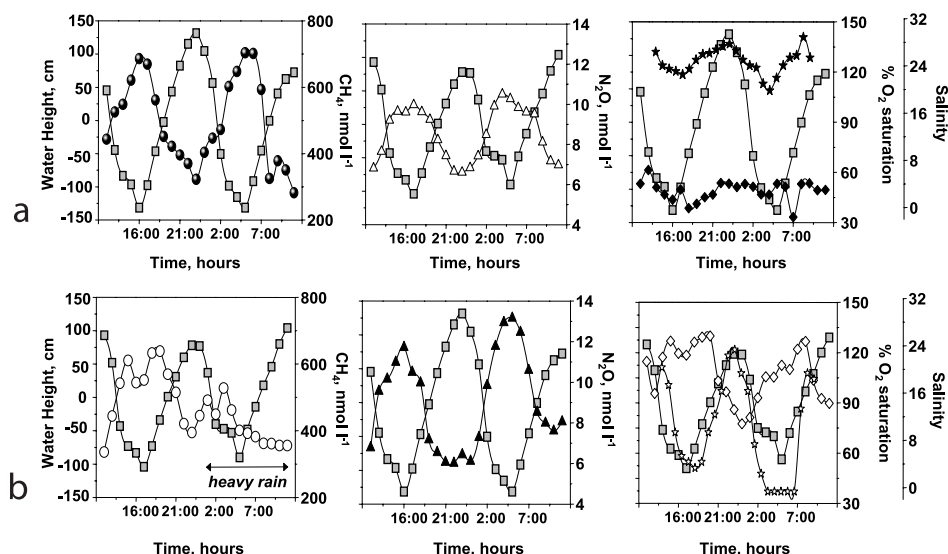
[5] Water samples (0.5 m) for CH<sub>4</sub> and N<sub>2</sub>O analyses were collected hourly from mid creek, inoculated with HgCl<sub>2</sub>, dark stored at 4°C for <6 hours, and analyzed by gas chromatography with a precision (1σ) of ±1% [Upstill-Goddard *et al.*, 1996]. Filtered samples (Whatman 0.4 μm) for NO<sub>3</sub><sup>-</sup>, NO<sub>2</sub><sup>-</sup> and NH<sub>4</sub><sup>+</sup> were stored frozen at -20°C and analyzed by standard colorimetry; analytical precisions (1σ) were better than ±2%. Salinity, temperature, turbidity and dissolved O<sub>2</sub> were measured on site with a portable probe (Horiba U-10); precisions (1σ) were ±0.01 salinity, ±3% turbidity, ±0.3°C, and ±0.01 mg O<sub>2</sub> l<sup>-1</sup>. Wind speed was recorded as one minute means with a hand held cup anemometer (Lutron LM8000; resolution ±0.1 m s<sup>-1</sup>).

[6] Emissions fluxes of CH<sub>4</sub> and N<sub>2</sub>O were measured directly from mangrove forest sediment (0.2 m<sup>3</sup> static chamber, both seasons) and surrounding creek waters (0.02 m<sup>3</sup> free-floating chamber, wet season) [Purvaja and Ramesh, 2001; Purvaja *et al.*, 2004]. For the static chamber a base collar inserted 10 cm into the sediment about 3 hours prior to sampling minimized lateral creek water inflow. Sampling commenced at sunrise low tide. Chamber sediment was water covered throughout both deployments; water depths increased from ~8 to ~15 cm during sampling. Chamber air was mixed with a D.C. pump and sampled every 20 minutes until the base collar detached from the sediment in the rising tide, giving ~3 hours of data. The free floating chamber was deployed in mid creek just after midnight high tide on 22 July 2004. Headspace was sampled every 30 min. for 3 hours. Deployment coincided with four hours of torrential rain from 0100 hours; about 6–8 cm of rain fell during this period. Creek water fluxes (both surveys) were also estimated using  $F =$

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**Figure 1.** Tidal variation of CH<sub>4</sub>, N<sub>2</sub>O, salinity and percent O<sub>2</sub> saturation in Wright Myo mangrove creek waters during (a), dry season (January 2004); (b), wet season (July 2004).

$k_w L \Delta p$ , where  $F$  is the gas flux ( $\text{mol m}^{-2} \text{d}^{-1}$ ),  $k_w$  is the gas transfer velocity ( $\text{cm hr}^{-1}$ ),  $L$  is gas solubility ( $\text{mol cm}^{-3} \text{atm}^{-1}$ ), and  $\Delta p$  is the water-to-air gas partial pressure difference. We derived  $k_w$  from a  $k_w$ -wind speed relation for CO<sub>2</sub> in estuaries [Clark *et al.*, 1995]. We chose this because it takes better account of non-wind speed driven turbulence (e.g., tidal currents) than do alternative relations [e.g., Liss and Merlivat, 1986; Wanninkhof, 1992].

### 3. Results and Discussion

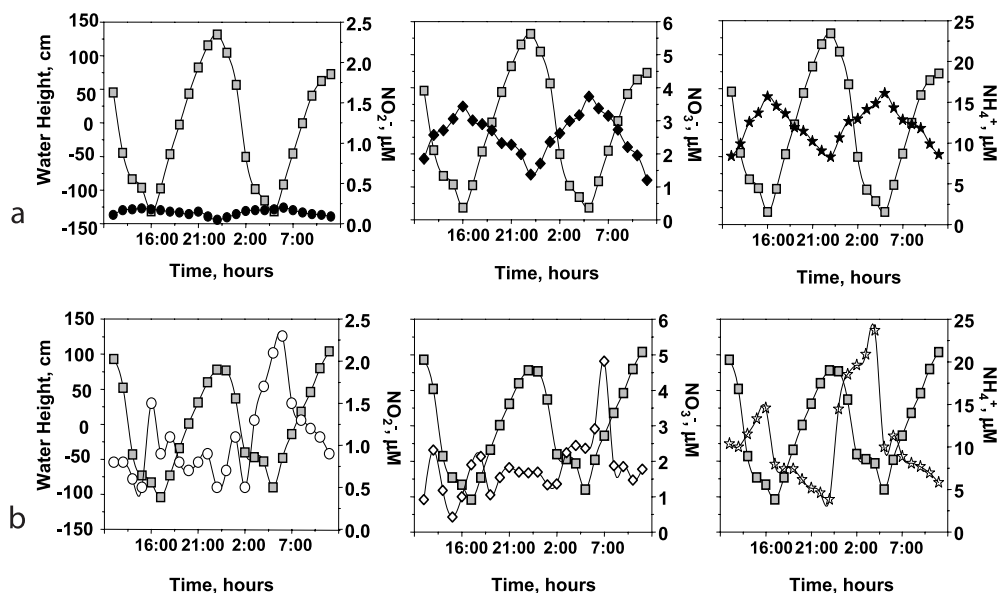
#### 3.1. CH<sub>4</sub>, N<sub>2</sub>O, O<sub>2</sub> and Nutrient Dynamics

[7] The means and ranges of dissolved CH<sub>4</sub> and N<sub>2</sub>O concentrations during the dry season (mean CH<sub>4</sub>  $491 \pm 133 \text{ nmol l}^{-1}$ , range 282–704  $\text{nmol l}^{-1}$ ; mean N<sub>2</sub>O  $9.0 \pm 2.3 \text{ nmol l}^{-1}$ , range 6.0–13.2  $\text{nmol l}^{-1}$ ) did not differ significantly from those during the wet season (mean CH<sub>4</sub>  $466 \pm 94 \text{ nmol l}^{-1}$ , range 324–625  $\text{nmol l}^{-1}$ ; mean N<sub>2</sub>O  $8.6 \pm 1.3 \text{ nmol l}^{-1}$ , range 6.6–10.5  $\text{nmol l}^{-1}$ ); no seasonal trend was discernable for either CH<sub>4</sub> or N<sub>2</sub>O (Figure 1). This is consistent with mean water temperatures of  $26.6 \pm 1.3^\circ\text{C}$  (dry season) and  $27.5 \pm 0.9^\circ\text{C}$  (wet season); any seasonal thermal signal in methanogenesis or nitrification-denitrification rates was therefore minimal. The CH<sub>4</sub> and N<sub>2</sub>O concentrations correspond to saturations  $\sim 12,000$ – $28,000\%$  and  $\sim 102$ – $208\%$  respectively; hence Wright Myo mangrove surrounding waters are a potentially strong CH<sub>4</sub> source but a weaker N<sub>2</sub>O source to air throughout the year.

[8] During both surveys, CH<sub>4</sub>, N<sub>2</sub>O, NH<sub>4</sub><sup>+</sup> and NO<sub>3</sub><sup>-</sup> were more strongly (negatively) correlated with tidal height ( $r^2$ : CH<sub>4</sub> = 0.84; N<sub>2</sub>O = 0.94) than with salinity ( $r^2$ : CH<sub>4</sub> = 0.80; N<sub>2</sub>O = 0.88) (all:  $p < 0.001$ ,  $n = 24$ ). Given the lack of seasonality in CH<sub>4</sub> and N<sub>2</sub>O concentrations and the contrasting salinity ranges (dry season 20–27; wet season 0–28; Figure 1), tidal dilution apparently only negligibly affects dissolved gas concentrations at Wright Myo. Dissolved O<sub>2</sub> gives insight into the controlling mechanisms (Figure 1). In the wet season (July) survey dissolved O<sub>2</sub> was

on average supersaturated (mean  $108 \pm 16\%$ , range 78–130%), whereas in the dry season (January) survey it was always undersaturated (mean  $49 \pm 6\%$ , range 41–61%). In both surveys percent O<sub>2</sub> saturation (%O<sub>2</sub>) deviated from that expected from diurnal primary production and respiration cycles (i.e., increase until sunset and then decrease until dawn). In July %O<sub>2</sub> was maximal prior to sunset (low tide). It initially decreased during the night but reached a minimum around high tide, 5–6 hours prior to sunrise (low tide). It then increased until about three hours after sunrise, after which it declined. In January %O<sub>2</sub> followed an opposite trend; it declined to a minimum around sunset (low tide), increased to a maximum around midnight (high tide), declined to a minimum around sunrise (low tide), and then increased again (Figure 1). These data clearly include a tidal signal. During January the decrease in %O<sub>2</sub> toward low tide and its increase toward high tide are consistent with tidal pumping in which low-O<sub>2</sub> porewaters seep into the creek waters from surrounding mangrove sediments as hydrostatic pressure gradually falls and then rises [Ovalle *et al.*, 1990]. Presumably during the July survey these effects were somewhat masked by high photosynthesis and respiration rates, and the torrential rain from around 0100 hours. During the rain salinity rapidly fell toward zero, much lower than during the previous low water, and it remained so for several hours (Figure 1). The increasing %O<sub>2</sub> between midnight high water and sunrise low water is not inconsistent with continued high night time respiration because the heavy rain could have effected significant reoxygenation, offsetting the impact of respiration on %O<sub>2</sub>. Tidal pumping of nutrients [Lara and Dittmar, 1999] and carbon system parameters [Borges *et al.*, 2003] has been observed in other tidal mangroves.

[9] The dry season survey (January) NH<sub>4</sub><sup>+</sup> contribution to total dissolved inorganic nitrogen (DIN) was high (Figure 2: mean  $82 \pm 2\%$ , range 79–87%) and similar to values in other mangrove ecosystems [Lara and Dittmar, 1999], whereas the wet season survey (July) NH<sub>4</sub><sup>+</sup> contribution was more variable and lower (Figure 2: mean  $77 \pm 10\%$ ,



**Figure 2.** Tidal variation of dissolved inorganic  $\text{NH}_4^+$ ,  $\text{NO}_3^-$  and  $\text{NO}_2^-$  in Wright Myo mangrove creek waters during (a), dry season (January 2004); (b), wet season (July 2004).

range 58–93%). The rainfall during this survey actively mobilized the surface sediment and DIN rapidly increased to about twice its mean value over the previous 12 hours; the  $\text{NH}_4^+$  fraction of DIN correspondingly increased to  $88 \pm 3\%$ . Both  $\text{NO}_2^-$  and  $\text{NO}_3^-$  also increased, but they lagged behind  $\text{NH}_4^+$  by about 2 hours (Figure 2). When the rain ceased DIN began to decline. The  $\text{NH}_4^+$  fraction of DIN also decreased but both  $\text{NO}_2^-$  and  $\text{NO}_3^-$  continued to increase before eventually declining toward their pre-rain values; the  $\text{NO}_2^-$  peak was about 2–3 hours ahead of the  $\text{NO}_3^-$  peak (Figure 2). This is consistent with nitrification of  $\text{NH}_4^+$  mobilized from the sediment porewaters, to  $\text{NO}_3^-$  via  $\text{NO}_2^-$ .

[10] During and after the rain the anti-correlation between  $\text{CH}_4$  and tidal height observed at other times was comparatively weak;  $\text{CH}_4$  concentrations were apparently suppressed whereas  $\text{N}_2\text{O}$  was apparently unaffected (Figure 1). This presumably indicates  $\text{CH}_4$  oxidation during sediment surface mobilization. Turbidity increased abruptly from  $<100 \text{ mg l}^{-1}$  over the first 15 hours of the survey to  $270 \pm 130 \text{ mg l}^{-1}$  during the rain; enhanced  $\text{CH}_4$  oxidation has been demonstrated following clay mineral addition to methanotroph cultures [Weaver and Dugan, 1972]. Such conditions would also favor  $\text{N}_2\text{O}$  production via nitrification of  $\text{NH}_4^+$  [Goreau et al., 1980], consistent with the observed behavior of  $\text{NO}_3^-$  and  $\text{NO}_2^-$  (Figure 2) If on the other hand the  $\text{N}_2\text{O}$  was mainly from denitrification it would have been suppressed by increased  $\text{O}_2$  during the rain, but there is no evidence for this (Figure 1). Other studies also indicate a predominantly nitrification source for mangrove  $\text{N}_2\text{O}$  [Bauza et al., 2002; Munoz-Hincapie et al., 2002].

### 3.2. $\text{CH}_4$ and $\text{N}_2\text{O}$ Emissions at Wright Myo

[11] Mangrove creek water  $\text{CH}_4$  and  $\text{N}_2\text{O}$  emissions derived from the Clark et al. [1995] relation (Table 1) are the means of hourly estimates over the full 24 hour surveys, whereas static (mangrove forest sediment-water-air emissions) and floating chamber (creek water emissions) data

(Table 1) were only collected for 2–3 hour periods. Chamber fluxes were therefore adjusted to take account of the strong diurnal signals in creek water  $\text{CH}_4$  and  $\text{N}_2\text{O}$ , in order to give representative mean fluxes for the full 24 hour surveys. We assumed the instantaneous water-to-air flux in each case to be proportional to the water-air partial pressure difference only; for the free-floating chamber this assumption is valid given that wind speeds were low and approximately constant during both seasons, hence flux variability related to short term changes in  $k_w$  can be discounted. For the static chamber it applies indirectly via the link between creek water gas content and tidal pumping due to hydrostatic pressure (tidal height); we assume that lateral creek water influx was minimal due to its inhibition by the chamber base collar. Mean creek water  $\text{N}_2\text{O}$  concentrations during static chamber deployments were  $\sim 10\%$  higher (dry season) and  $\sim 5\%$  higher (wet season) than over the full tidal cycles; to convert to 24-hour fluxes the chamber fluxes were adjusted downward by these amounts. Similarly, the dry season  $\text{CH}_4$  flux was adjusted downward by  $\sim 13\%$ , whereas the wet season flux was adjusted upward by  $\sim 30\%$  due to the rain-induced decline in creek water  $\text{CH}_4$  during deployment (Figure 1). We consider the precipitation event to have been atypical, hence our 24-hour estimate represents a wet season “dry” day. Floating chamber fluxes were

**Table 1.**  $\text{CH}_4$  and  $\text{N}_2\text{O}$  Emission Fluxes ( $\mu\text{mol m}^{-2} \text{ hr}^{-1}$ ) at Wright Myo; Observations During January 2004 (Dry Season) and July 2004 (Wet Season)

	Mangrove Creek		Mangrove Forest	
	Wet Season	Dry Season	Wet Season <sup>a</sup>	Dry Season <sup>a</sup>
$\text{N}_2\text{O}$	1.3 <sup>b</sup>	0.23 <sup>c</sup>	0.24 <sup>c</sup>	0.21
$\text{CH}_4$	34.5 <sup>b</sup>	23.0 <sup>c</sup>	23.9 <sup>c</sup>	17.8

<sup>a</sup>Static chamber estimates.

<sup>b</sup>Floating chamber estimates.

<sup>c</sup>Estimated from the Clark et al. [1995] relation.



**Table 2.** CH<sub>4</sub> and N<sub>2</sub>O Emission Fluxes (μmol m<sup>-2</sup> hr<sup>-1</sup>) From Mangrove Ecosystems<sup>a</sup>

Site	Location	Emission, μmol m <sup>-2</sup> hr <sup>-1</sup>		Reference
		N <sub>2</sub> O	CH <sub>4</sub>	
Queensland, Australia		(-0.1)–0.3	1–22	<i>Kreuzwieser et al.</i> [2003]
NE Hainan I. China	(~19°N 110°E)		0.9–2.8	<i>Lu et al.</i> [1999]
Jiulongjiang Estuary, China	(~24°N 117°E)	0–200	0–100	<i>Alongi et al.</i> [2005]
SW India	(~11–13°N 79°E)		125–813	<i>Purvaja and Ramesh</i> [2001]
Pichavarum/Cauvery Deltas, SW India	(~11°N 79°E)		21–67	<i>Purvaja et al.</i> [2004]
SW Puerto Rico	(~18°N 67°W)		10–215	<i>Sotomayor et al.</i> [1994]
SW Puerto Rico	(~18°N 67°W)	0.1–7.8		<i>Corredor et al.</i> [1999]
SW Puerto Rico	(~18°N 67°W)	0.05–1.4		<i>Bauza et al.</i> [2002]
SW Puerto Rico	(~18°N 67°W)	0.05		<i>Munoz-Hincapie et al.</i> [2002]
Mtoni Mangrove, Tanzania	(~6°S 39°E)		(-0.8)–520	<i>Lyimo et al.</i> [2002]
Ranong Province, Thailand	(~10°N 98°E)		12–33	<i>Lekphet et al.</i> [2005]
Wright Myo, Andaman Is	(~12°N 93°E)	0.1–1.3	18–35	This Study

<sup>a</sup>All fluxes excepting some of those in this work were estimated using static chambers; the negative emissions in parentheses indicate a net flux into the mangrove sediment.

corrected based on creek water data for the first 12 hours of the survey only, ignoring the post-rain data; for both gases the required adjustments were ~10%. For N<sub>2</sub>O, which was apparently not measurably affected by the rain (Figure 1), adjusting the floating chamber flux over the full 24 hours did not change this outcome. The discrepancy in required adjustments between the static and floating chambers relates to their being deployed at different times.

[12] During both surveys we observed periodic bubble ebullition, which was most intense around low tide. Hydrostatic pressure changes of a few percent can significantly increase ebullition [*Ostrovsky*, 2003], and increased sediment shear stress also contributes [*Joyce and Jewell*, 2003]. CH<sub>4</sub> ebullition can exceed diffusional/turbulent exchange in shallow systems by more than an order of magnitude [*Ostrovsky*, 2003], and it can be significant for other gases via diffusion into the bubbles [*Leifer and Patro*, 2002]. The flux estimates (Table 1) therefore likely represent minimum values.

[13] Creek water fluxes estimated with the *Clark et al.* [1995] relation showed no seasonality, consistent with the concentration data (Figure 1) and the insensitivity of  $k_w$  to the low ambient wind speed ranges encountered (wet season,  $4.2 \pm 1.0$  m s<sup>-1</sup>; dry season  $2.0 \pm 0.5$  m s<sup>-1</sup>). At these wind speeds alternative widely used relations underestimate [*Liss and Merlivat*, 1986] or overestimate [*Wanninkhof*, 1992]  $k_w$  measured in tidal estuaries with gaseous tracers [*Clark et al.*, 1995]. CH<sub>4</sub> emissions were about 100 times higher than those of N<sub>2</sub>O. Creek water fluxes determined with the floating chamber were 1.5 times higher for CH<sub>4</sub> and ~5.5 times higher for N<sub>2</sub>O than these wind speed based estimates (Table 2). Some authors have found floating chambers to overestimate gas exchange relative to other methods [*Clark et al.*, 1995; *Raymond and Cole*, 2001], however others find much closer agreement [*Kremer et al.*, 2003]. Although the intense rain during deployment could have suppressed wind waves at the low wind speeds encountered, strong rain-enhancement of gas exchange via increased surface turbulence and surface bubble entrainment is well known [*Ho et al.*, 2000]. Bubble entrainment increases with rain intensity and drop size but overall surface turbulence effects dominate, accounting for about 80% of the total enhancement for low solubility gases [*Ho et al.*, 2000]. Based on our rainfall estimate and laboratory data [*Ho et al.*, 2000]  $k_w$  enhancement of the

order observed is not unlikely. Given the small rain-enhancement of CH<sub>4</sub> exchange relative to N<sub>2</sub>O, this supports our hypothesis of creek water CH<sub>4</sub> oxidation and the corresponding changes to DIN speciation during intense rainfall (Figure 1). Mangrove forest sediment CH<sub>4</sub> fluxes are also consistent with this hypothesis, being ~40% lower during the wet season survey (Table 2). In contrast the wet season survey forest sediment N<sub>2</sub>O flux was about twice as high as the dry season survey flux. This may reflect additional N<sub>2</sub>O production in the sediments via the nitrification of NH<sub>4</sub><sup>+</sup>, as implied by the DIN data (Figure 1).

[14] There are few available estimates of N<sub>2</sub>O or CH<sub>4</sub> emissions from mangrove ecosystems (Table 2) and none of these considers the contribution from surrounding creek waters, even though at Wright Myo this is comparable per unit area to the flux from mangrove sediment (Table 1). Our N<sub>2</sub>O and CH<sub>4</sub> emissions estimates for Wright Myo are toward the low end of previously published ranges for mangroves (Table 2). These span two orders of magnitude; however the data include both anthropogenically impacted and pristine sites. For example, the Jiulongjiang estuary, China, [*Alongi et al.*, 2005] is highly urbanized, high CH<sub>4</sub> emissions for SW Puerto Rico [*Sotomayor et al.*, 1994] and SE India [*Purvaja and Ramesh*, 2001] reflect domestic sewage pollution, and the N<sub>2</sub>O emissions for SW Puerto Rico reflect anthropogenic N addition [*Corredor et al.*, 1999]. Overall mangrove ecosystems appear to be a larger source of CH<sub>4</sub> than N<sub>2</sub>O to the atmosphere (Table 2).

### 3.3. Global Fluxes of CH<sub>4</sub> and N<sub>2</sub>O from Mangrove Ecosystems

[15] Given the relatively low CH<sub>4</sub> and N<sub>2</sub>O emissions from pristine Wright Myo (Table 2), up scaling these based on estimated global areas of mangrove forest and creek enables a conservative estimate of the CH<sub>4</sub> and N<sub>2</sub>O source strengths for mangrove ecosystem worldwide, the first such estimate to include the contributions from both mangrove forest and surrounding creek waters. For this we assume worldwide mangrove forest and creek areas ~0.2 × 10<sup>6</sup> km<sup>2</sup> and ~0.36 × 10<sup>6</sup> km<sup>2</sup> respectively [*Borges et al.*, 2003; *Selvam*, 2003], and similar wet and dry season floating chamber fluxes based on measured creek water concentrations and wind speeds. Worldwide mangrove ecosystem emissions to the atmosphere are thus ~13 × 10<sup>10</sup> mol CH<sub>4</sub> yr<sup>-1</sup> ( $2.2 \times 10^{12}$  g yr<sup>-1</sup>) and 2.7 ×

$10^9$  mol  $\text{N}_2\text{O}$   $\text{yr}^{-1}$  ( $1.2 \times 10^{11}$  g  $\text{N}_2\text{O}$   $\text{yr}^{-1}$ ). Previously estimated worldwide estuarine emissions are  $\sim 5\text{--}8 \times 10^{10}$  mol  $\text{CH}_4$   $\text{yr}^{-1}$  and  $\sim 8\text{--}13 \times 10^{10}$  mol  $\text{N}_2\text{O}$   $\text{yr}^{-1}$  [Bange *et al.*, 1994, 1996]. Even though our emission flux estimates for Wright Myo are conservative relative to data from other mangroves (Table 2) and exclude the potential ebullition contribution for  $\text{CH}_4$ , they nevertheless imply that  $\text{CH}_4$  fluxes from mangrove ecosystems could dominate  $\text{CH}_4$  emissions from coastal ecosystems worldwide, and that previous work may have seriously underestimated the magnitude of this flux. By contrast, mangrove ecosystems appear to be relatively small contributors to the net global flux of atmospheric  $\text{N}_2\text{O}$  from coastal sites.

[16] It is instructive to examine the potential net greenhouse gas budget of mangrove ecosystems by comparing our  $\text{CH}_4$  and  $\text{N}_2\text{O}$  flux estimates with the available data for  $\text{CO}_2$ . There are currently no simultaneously derived  $\text{CO}_2$  flux estimates for mangrove forest and creek waters, and no simultaneous measurements of  $\text{CO}_2$ ,  $\text{CH}_4$  and  $\text{N}_2\text{O}$  fluxes for either environment. The available  $\text{CO}_2$  data for creek waters imply net heterotrophy and an atmospheric  $\text{CO}_2$  source  $\sim 50 \times 10^{12}$  g C  $\text{y}^{-1}$ ; this is based on a creek water area  $\sim 0.2 \times 10^6$  km<sup>2</sup> [Borges *et al.*, 2003] which may be an underestimate [Selvam, 2003]. Even so, assuming that  $\sim 75\%$  of the total carbon flux into mangrove ecosystems is respired and re-emitted as  $\text{CO}_2$  with only  $\sim 2\text{--}3\%$  carbon burial in sediments, globally mangrove ecosystems (forest and creek) may be a small net sink for atmospheric  $\text{CO}_2$   $\sim 18 \times 10^{12}$  g C  $\text{y}^{-1}$  [Alongi, 2002]. Based on this and taking into account means of 20-year and 100-year global warming potentials of 286 for  $\text{N}_2\text{O}$  and 43 for  $\text{CH}_4$  [Intergovernmental Panel on Climate Change, 2001], mangroves could be an overall small greenhouse gas source to the troposphere equivalent to  $\sim 100 \times 10^{12}$  g  $\text{CO}_2$   $\text{y}^{-1}$ . However, uncertainty in this estimate is inherently large because source data remain very scarce.

[17] New data for  $\text{N}_2\text{O}$  and  $\text{CH}_4$  reported here raise an important question: what is the overall role of mangrove ecosystems in the greenhouse gas budget of the troposphere? The inherent uncertainty of our estimate herein derived highlights a need for further detailed study. Although the direct addition of carbon and nitrogen to mangrove ecosystems demonstrably leads to enhanced emissions of  $\text{N}_2\text{O}$  and  $\text{CH}_4$  [Purvaja and Ramesh, 2001; Munoz-Hincapie *et al.*, 2002], wholesale mangrove clearance on the other hand may result in net heterotrophy and accelerated emissions of  $\text{CO}_2$  [Alongi *et al.*, 2004]. Understanding and evaluating the balance of these processes is highly topical, being relevant to evaluating the efficacy of proposed  $\text{CO}_2$  mitigation strategies that involve mangrove replanting, for example. Additional data are therefore required from a range of mangrove ecosystems to better constrain the net consequences of these effects. Future studies could usefully focus on comparing emissions from similar sites with and without mangroves, emissions before and after mangrove plantation, and emissions from established and eroding mangrove sites.

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