Denitrification and Anammox in Tropical Aquaculture Settlement Ponds: An Isotope Tracer Approach for Evaluating N2 Production

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Abstract

Settlement ponds are used to treat aquaculture discharge water by removing nutrients through physical (settling) and biological (microbial transformation) processes. Nutrient removal through settling has been quantified, however, the occurrence of, and potential for microbial nitrogen (N) removal is largely unknown in these systems. Therefore, isotope tracer techniques were used to measure potential rates of denitrification and anaerobic ammonium oxidation (anammox) in the sediment of settlement ponds in tropical aquaculture systems. Dinitrogen gas (N2) was produced in all ponds, although potential rates were low (0–7.07 nmol N cm⁻³ h⁻¹) relative to other aquatic systems. Denitrification was the main driver of N₂ production, with anammox only detected in two of the four ponds. No correlations were detected between the measured sediment variables (total organic carbon, total nitrogen, iron, manganese, sulphur and phosphorous) and denitrification or anammox. Furthermore, denitrification was not carbon limited as the addition of particulate organic matter (paired t-Test; P = 0.350, n = 3) or methanol (paired t-Test; P = 0.744, n = 3) did not stimulate production of N₂. A simple mass balance model showed that only 2.5% of added fixed N was removed in the studied settlement ponds through the denitrification and anammox processes. It is recommended that settlement ponds be used in conjunction with additional technologies (i.e. constructed wetlands or biological reactors) to enhance N₂ production and N removal from aquaculture wastewater.

Introduction

The release of anthropogenic N to the coastal zone poses a threat to many shallow marine ecosystems [1]. Discharge of aquaculture wastewaters has contributed to N enrichment of some coastal regions [2] and settlement ponds have been established as a remediation strategy from aquaculture wastewater prior to release to the environment [3,4]. Settlement pond technologies are widely implemented as a low cost option for treating municipal [5], fish farm [6] and dairy farm wastewater [7]. However, the nutrient removal efficiency of settlement ponds associated with land-based tropical aquaculture systems is unclear. Generally, newly established (<1 yr old) settlement ponds, with a basic design, provide significant reductions in total suspended solids, but are less efficient in the remediation of dissolved nutrients [3,8]. Furthermore, given that the efficiency of wetland wastewater treatment systems can decrease with age [9], it is likely that the performance of settlement ponds, which act as brackish water constructed wetlands, will decrease over time unless they are actively managed. Methods to improve the long term performance of tropical aquaculture settlement ponds include the use of extractive organisms such as algae, which can be cultured and subsequently harvested [10], and also the removal of settled organic rich particulates (sludge) which prevents remineralization of dissolved N back into the water column [5,11]. Microbial nutrient transformation, which is largely un-quantified, also presents a potentially significant mechanism to reduce dissolved inorganic nitrogen (DIN) in aquaculture wastewater.

Denitrification and anammox are the major microbial processes removing fixed N from wastewater through the production of dinitrogen gas (N₂). During denitrification, nitrate (NO₃⁻) is reduced to nitrite (NO₂⁻), nitric oxide (NO) and nitrous oxide (N₂O), before eventually being converted to N₂. Anammox also directly removes fixed N and couples NO₂⁻ reduction with ammonium (NH₄⁺) oxidation to produce N₂ [12,13]. Denitrification and anammox are also important for the removal of N from natural system such as intertidal flats [14], marsh sediments [15], deep anoxic waters [16] and sediments from the continental shelf (50 m) and slope (2000 m) [17]. Denitrification and anammox in natural systems can remove up to 266 mmol m⁻² d⁻¹ and 61 mmol m⁻² d⁻¹ of N, respectively [16]. These processes may be active in the treatment of aquaculture effluent water and could be exploited to enhance treatment. However, to date there has
been no published quantification of denitrification and anammox in settlement pond systems treating waste from tropical aquaculture farms.

The first step in optimizing the removal of fixed N through the denitrification and anammox pathways is to quantify their activity in settlement ponds and relate this to the environment of the ponds. Accordingly, the aim of this study was to determine if denitrification and anammox occur in sediments collected from tropical settlement ponds that are used to treat effluent from commercial production of prawns (shrimp) and fish. We used sediment slurry assays to investigate potential N2 production in multiple zones of four settlement ponds on three farms (two prawn farms and one fish farm). We also investigated the relationship between the potential rates of N2 production with the geochemical characteristics of the ponds. Additionally, the effect of carbon on N2 production was tested since intensive aquaculture systems have N rich wastewaters where microbial N removal is typically limited by the supply of carbon as an electron donor [18]. Together these data provide new insight into N cycling processes in shallow tropical eutrophic marine systems in the context of N management.

Methods

Study site

The presence of denitrification and anammox and their potential rates were measured in sediment collected from four settlement ponds across two operational prawn (Penaeus monodon) farms and one barramundi (Lates calcarifer Bloch) farm. At Farm 1 sediment was collected from the two functional settlement ponds, this allowed comparison of N2 production over small spatial scales (A and B; Figure 1). Additionally, sediment was collected from the only settlement pond at Farm 2 (Pond C) and the only settlement pond at Farm 3 (Pond D) (Figure 1). The three farms spanned the wet and dry tropics allowing comparison of N2 production in different environments. Each pond was split into 3 zones (Z1, Z2 and Z3) (Figure 1). In all ponds Z1 was near the inlet, Z2 was near the middle of the settlement pond, and Z3 was near the outlet of the settlement pond. Ponds have diurnal fluctuations in dissolved oxygen (DO) concentration; from <31.2 μM at night to supersaturation (>312.5 μM) during the day, indicating rapid water column productivity. Similarly, there are diurnal pH fluctuations (1–1.5 pH). According to farm records, salinity fluctuates seasonally, with dramatic decreases from 35% to 5% caused by heavy precipitation over the summer wet season. During the wet season access to the farms by road is limited. All assays were modified from Erler et al. [20]. Sacrifice of the slurry samples involved the addition of 2 mL He headspace, capped without headspace and homogenized by inverting 2–3 times. Triplicate samples were sacrificed from each treatment at 0, 0.5, 17 h and 24 h by introducing 200 μL 50% w/v ZnCl2 through a rubber septum (n = 3). The 0 and 0.5 h time periods were chosen based on rapid turnover rates determined by Trimmer et al. [23] and 17 and 24 h were modified from Erler et al. [20]. Sacrifice of the slurry samples involved the addition of 2 mL He headspace to the samples through the septum. Samples were stored inverted and submerged in water at 4°C until analysis to ensure there was no diffusion of N2 into or out of the Exetainers. A gas chromatograph (Thermo Trace Ultra GC) interfaced to an isotope ratio mass spectrometer (IRMS, Thermo Delta V Plus IRMS) was used to determine 29N2 and 30N2 content of dissolved nitrogenous gas (includes 15N-NH4+ or 100 μM 15N-NO3− or 100 μM 15N-NH4+ plus 100 μM 15N-NO3−) were added to the slurries. After the isotope amendment, the Exetainers were filled with the degassed seawater, capped without headspace and homogenized by inverting 2–3 times. Triplicate samples were sacrificed from each treatment at 0, 0.5, 17 h and 24 h by introducing 200 μL 50% w/v ZnCl2 through a rubber septum (n = 3). The 0 and 0.5 h time periods were chosen based on rapid turnover rates determined by Trimmer et al. [23] and 17 and 24 h were modified from Erler et al. [20]. Sacrifice of the slurry samples involved the addition of 2 mL He headspace to the samples through the septum. Samples were stored inverted and submerged in water at 4°C until analysis to ensure there was no diffusion of N2 into or out of the Exetainers. A gas chromatograph (Thermo Trace Ultra GC) interfaced to an isotope ratio mass spectrometer (IRMS, Thermo Delta V Plus IRMS) was used to determine 29N2 and 30N2 content of dissolved nitrogenous gas (includes 15N-N2 and 15N-N2O, collectively referred to as N2). Varying volumes (3–10 L) of air were used as calibration standards.

The rate of N2 production in the 24 h incubation trials (above) was calculated from the slope of the regression over the incubation period (0, 0.5, 17, 24 h) based on Dalsgaard and Thamdrup [24]. However, in some cases the production of 15N2 and 15N2 was non-linear and rates were calculated based on the first two production points. Therefore a subsequent slurry assay was run to investigate N2 production rates over short, regular time intervals (15 min) to
gain a more accurate insight into potential process rates. Sediment for the additional assays was collected from settlement Pond D, Zones 1 (n = 1) and 3 (n = 1) in October 2010. These zones were chosen because the production of N₂ was non-linear during the 24 h incubation assay (see results section). Assays were run as described above, following the same sediment collection, pre-incubation, amendment and analysis techniques. However, samples were sacrificed at 0, 15, 30, 45 and 90 min.

Slurry assay with carbon manipulation

The effect of an additional carbon source on the occurrence of denitrification and anammox was tested with a separate set of slurry assays because organic carbon limits N₂ production in some aquaculture systems [25]. Extra sediment was collected in March and August (2010) in the sampling described above. Sediments from Ponds A (August) and C (March) were assayed with and without addition of a carbon source because organic carbon has stimulated or correlated with N₂ production in some systems previously [17,26,27]. Concentrated particulate organic matter (POM) was used to test the effect of an in situ carbon source collected from Pond A. POM was collected by transporting settlement pond-influent water to the laboratory at the same time that sediments were collected. Suspended solids in influent water were concentrated by centrifugation (10 min at 3000 rpm). 400 µL aliquots of concentrated (~100 mg L⁻¹) POM were added to Exetainer vials prior to the addition of amendments. However, in the absence of a high total suspended solid load at Pond C, methanol (MeOH) was used as the carbon source as it stimulates denitrification but inhibits anammox in some circumstances [28,29]. MeOH additions were carried out by adding MeOH at a concentration of 3 mM (based on Jensen et al. [29]) to a parallel set of samples from Pond C prior to amendments.

Modeling N removal

A simplistic model was constructed to estimate the mean dry season N removal (NR) capacity (%) of the four settlement ponds. NR was estimated using the potential N₂ production rates calculated in the present study, and N inputs into the pond through the wastewater. Given the substantial contribution of N remineralized from sludge in shrimp grow-out ponds (often exceeding inputs of N originating from feeds [30]), a variable to account for remineralization inputs was also added (N_{min}). The following equation was used to calculate N removal and the parameters are further defined in Table 1:

Equation 1.

\[
NR = \frac{N₂ \times A \times t \times A_r}{(N_{new} + N_{min})} \times 100
\]

where N₂ = the mean total (inclusive of anammox) N₂ production rate measured during the 24 h incubation (nmol N cm⁻² h⁻¹; Table 1). We adopted a conservative approach and assumed that N₂ production, driven by denitrification, only occurs in the top 1 cm of the sediment. Denitrification occurs at the oxic-anoxic
interface so the depth at which it occurs is dependent on O2 penetration into the sediments. O2 penetration is estimated at <0.5 mm in fish farm wastewater treatment ponds [4], 1.5–4 mm in sediments below fish cages and associated reference sites and up to 20 mm in a muddy macrotidal estuary [31]. This active zone is subsequently extrapolated to estimate rates for the entire area of the settlement pond. The remaining parameters are defined as follows: \( A \) = mean area of the settlement pond \( \left( \text{m}^2 \right) \); \( t = 24 \left( \text{h d}^{-1} \right) \); \( A \) = atomic weight of N; \( N_{\text{mean}} \) = mean rate of N input (inclusive of particulates and dissolved) via the wastewater (environmental protection agency (EPA) monitoring data, quantified monthly by Farm 1; kg N d\(^{-1}\)); \( N_{\text{mean}} \) = mean rate of N input via mineralisation (deduced from NH4\(^+\) and DON fluxes in Burford and Longmore [32]; Table 1; kg N d\(^{-1}\)).

### Calculations and statistical analysis

The sediment characteristics data was analysed as a 2-factor nested design, pond and zone (pond) using permutational multivariate analysis of variance (PERMANOVA) [33]. PERMANOVA calculated \( p \)-values from 9999 permutations based on Bray-Curtis distances. A 1-factor PERMANOVA was subsequently used to compare differences in \( N_2 \) production rate data (three variables: denitrification, anammox and total \( N_2 \) production) between ponds with zones as replicates \( (n = 3) \); 9999 permutations were again used to calculate \( p \)-values based on Bray-Curtis distance. PRIMER version 6 and PERMANOVA+ version 1.0.4 were used to conduct both analyses.

The relationship between \( N_2 \) production rate (three variables: denitrification, anammox and total \( N_2 \) production) and sediment characteristics was subsequently investigated using the BIOENV procedure in PRIMER. This procedure performs a rank correlation of the two similarity matrices (described above) and tests every combination of sediment characteristics to determine which set of variables best explains the observed \( N_2 \) production rates [34]. A Bray-Curtis similarity matrix comprised of both \( N_2 \) production rate data and sediment variable data was also used to conduct a hierarchical agglomerative cluster analysis which was superimposed on a multidimensional scaling (nMDS) plot. The nMDS plot provided a 2-D visualization of the relationship between sediment characteristics and \( N_2 \) production rates.

The effect of carbon addition on potential \( N_2 \) production rate in sediments was analyzed with paired \( t \)-Tests for each carbon source (POM and MeOH).

### Results

#### Pond characteristics and abiotic factors

Surface water temperature (25.8 \( \pm \)1.0\(^\circ\)C) and pH (7.6 \( \pm \)0.2) varied little similar across all ponds and zones. Surface water salinity in Pond C (Farm 2) was lower (17–18\%) than the other three ponds (31–35\%; Table 2) due to its location in the wet tropics where precipitation is high (Figure 1).

Sediment at all zones was uniformly dark black with minor color variation shown in a narrow lighter band (~3 mm oxic zone) at the surface of the sediment. The porosity ranged between 41–72\% (Table 2) and sediments produced a rich hydrogen-sulfide smell and gaseous bubbles (presumably consisting of a mix of biogases) at the water surface when the sediment was disturbed. Very little bioturbation by burrowing organisms or flora was evident. There was significant variability between ponds (Table 3; PERMANOVA; pond; Pseudo \( F \)= 2.06, P = 0.028) and between zones within ponds (Table 3; PERMANOVA; zone (pond); Pseudo \( F \)= 33.83, P<0.001). The variance in sediment characteristics at the finer scale (i.e. meters) between zones within ponds (52.4\%) was greater than the variance between settlement ponds located kilometers apart (31.6\%).

#### Denitrification and anammox potential

There was also a significant difference in the potential rate of \( N_2 \) production between ponds (Table 3; PERMANOVA; pond; Pseudo \( F \)= 3.91, P = 0.001). The potential rate was highest in sediments collected from pond A, with denitrification the sole producer of \( N_2 \) (7.07 \( \pm \)2.99 nmol N cm\(^{-3} \)\text{h}^{-1}; Table 4) and lowest in sediments collected from pond C, where again denitrification was the responsible for 100\% of the \( N_2 \) produced (0.004 \( \pm \)0.003 nmol N cm\(^{-3} \)\text{h}^{-1}; Table 4). However, there was no correlation between the potential production of \( N_2 \) in zones within ponds and different sediment characteristics that defined each pond (nMDS, Figure 2a & b). For example, pond B zone 3 had the highest anammox rates and low denitrification, whereas pond A, zones 2 and 3 had the opposite trend (Figure 2a). This is highlighted in the vector loadings for which the vectors for anammox and denitrification are clearly negatively correlated (Figure 2b).

### Tables

#### Table 1. An estimate of nitrogen inputs and microbial removal from settlement ponds, note TN = total nitrogen, WW = wastewater, min = mineralization.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Value</th>
<th>Unit</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pond area</td>
<td>6000</td>
<td>m(^2)</td>
<td>Farm proprietors Pers. comm.</td>
</tr>
<tr>
<td>Mean TN WW input</td>
<td>14.8</td>
<td>kg N d(^{-1})</td>
<td>EPA monitoring data</td>
</tr>
<tr>
<td>Mean net NH(_4^+) min</td>
<td>27.8</td>
<td>mmol m(^{-3}) h(^{-1})</td>
<td>[30]</td>
</tr>
<tr>
<td>Mean net DON min</td>
<td>0.6</td>
<td>mmol m(^{-3}) h(^{-1})</td>
<td>[30]</td>
</tr>
<tr>
<td>Mean ( N_2 ) production</td>
<td>2.9</td>
<td>mmol N cm(^{-3}) h(^{-1})</td>
<td>Slurry assay</td>
</tr>
<tr>
<td>Net N removal</td>
<td>2.5</td>
<td>%</td>
<td>Model</td>
</tr>
</tbody>
</table>

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#### Table 2. Mean surface water salinity \((n=3 \pm 1 \text{ SE})\) and abiotic sediment characteristics \((n=9 \pm 1 \text{ SE})\) in the four settlement ponds \((A, B, C \text{ and } D)\) used to treat aquaculture wastewater \((\mumol \text{ g}^{-1} \text { unless stated})\).

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Pond A</th>
<th>Pond B</th>
<th>Pond C</th>
<th>Pond D</th>
</tr>
</thead>
<tbody>
<tr>
<td>Salinity (%)</td>
<td>31±0</td>
<td>34±0</td>
<td>18±0</td>
<td>35±0</td>
</tr>
<tr>
<td>Porosity (%)</td>
<td>0.5±0.0</td>
<td>0.5±0.0</td>
<td>0.5±0.0</td>
<td>0.5±0.0</td>
</tr>
<tr>
<td>TOC (%)</td>
<td>61±13</td>
<td>62±6</td>
<td>43±5</td>
<td>63±4</td>
</tr>
<tr>
<td>TOC (%)</td>
<td>0.7±0.9</td>
<td>0.8±0.1</td>
<td>0.5±0.1</td>
<td>0.8±0.1</td>
</tr>
<tr>
<td>TN</td>
<td>5±1</td>
<td>6±1</td>
<td>4±1</td>
<td>8±1</td>
</tr>
<tr>
<td>TN (%)</td>
<td>0.1±1.0</td>
<td>0.1±0.4</td>
<td>0.1±0.8</td>
<td>0.1±0.6</td>
</tr>
<tr>
<td>TP</td>
<td>18±4</td>
<td>14±2</td>
<td>5±1</td>
<td>14±3</td>
</tr>
<tr>
<td>Fe</td>
<td>9±1</td>
<td>9±2</td>
<td>12±2</td>
<td>9±0</td>
</tr>
<tr>
<td>Mn</td>
<td>8±2</td>
<td>6±1</td>
<td>1±0</td>
<td>2±0</td>
</tr>
</tbody>
</table>

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loadings of all the sediment characteristics. Anammox did cluster near sediment variables (Figure 2b), however there was no correlation between the N2 production matrix (inclusive of total N2 denitrification and anammox) or the sediment variable matrix (BIOENV analysis; \( r = 0.134, P = 0.730 \)).

In incubations with \(^{15}N\) labeling of nitrate only, the majority of \(^{15}N\)-NO\(_3\) converted to N\(_2\) was found in \(^{36}N\)N\(_2\) (Figure 3). Only in pond B was more of \(^{15}N\)-NO\(_3\) that was converted to N\(_2\) found in \(^{29}N\)N\(_2\) than in \(^{36}N\)N\(_2\) (Figure 3). Anammox was detected in pond B sediments as indicated by the higher percent recovery (0.67\(\pm\)0.28\%) of \(^{15}N\)-N\(_2\) in treatments where \(^{15}N\)-NH\(_4\)\(^+\) and unlabelled \(^{14}N\)-NO\(_3\) were added compared to treatments where \(^{15}N\)-NH\(_4\)\(^+\) was added (0.28\(\pm\)0.09\%); Table 5). However, in this pond total recovery of \(^{15}N\)-NO\(_3\) as \(^{15}N\)-N\(_2\) was extremely low (0.20\(\pm\)0.07; Table 5).

**Table 3.** A summary of statistical analyses; PERMANOVAs based on the Bray-Curtis similarities of transformed (4th root) sediment characteristic data and potential N\(_2\) production rate data.

<table>
<thead>
<tr>
<th>Sediment characteristics</th>
<th>Test</th>
<th>( \text{df} )</th>
<th>( MS )</th>
<th>( P )</th>
</tr>
</thead>
<tbody>
<tr>
<td>Factors</td>
<td>PERMANOVA</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Pond</td>
<td>3</td>
<td>39</td>
<td>2.06</td>
<td>0.028</td>
</tr>
<tr>
<td>Zone (Pond)</td>
<td>8</td>
<td>19</td>
<td>33.83</td>
<td>0.000</td>
</tr>
</tbody>
</table>

Discussion

**Total N\(_2\) production and controlling mechanisms**

Isotope tracer techniques confirmed the production of N\(_2\) in sediment collected at all three zones within each of the four settlement ponds used to treat wastewater from commercial prawn and barramundi farms. The potential rates (0–7.07 nmol N cm\(^{-3}\) h\(^{-1}\)) were within the range of those reported for a subtropical constructed wetland (1.1\(\pm\)0.2 to 13.1\(\pm\)2.6 nmol N cm\(^{-3}\) h\(^{-1}\)) [20], but lower than those reported for subtropical mangrove and shrimp grow out pond sediments (21.5–78.5 nmol N cm\(^{-3}\) h\(^{-1}\)) [35]. Nevertheless, it can be assumed that both denitrifying bacteria and *Planctomycetes* (anammox bacteria) are present in the ponds and that there is potential to stimulate N\(_2\) production rates and enhance N removal. To achieve this, an understanding of the mechanisms controlling N\(_2\) production is required. We therefore investigated the effect of carbon additions on N\(_2\) production rate and the relationship between the concentration of sediment elements and N\(_2\) production rates. However, there was no significant change in the rate of N\(_2\) production under carbon loading and there was no correlation between any of the measured sediment variables and N\(_2\) production rate via denitrification or anammox.

Denitrification is often limited by carbon in aquaculture ponds, as carnivorous marine species require high inputs of protein rich feeds. N removal can be enhanced through the addition of an exogenous carbon source, for example glucose and cassava meal [26] or molasses [25] have been added to shrimp farm wastewater treatment processes, resulting in up to 99% removal of NH\(_4\)\(^+\), NO\(_3\)\(^-\) and NO\(_2\)\(^-\). Similarly, methanol is a common additive to enhance denitrification for municipal wastewater treatment, increasing degradation of NO\(_2\)\(^-\) in activated sludge from 0.27 mg NO\(_2\)\(^-\) g\(^{-1}\) volatile suspended solids (VSS) h\(^{-1}\) to 1.20 mg NO\(_2\)\(^-\) g\(^{-1}\) VSS h\(^{-1}\) [27]. However, in the present study N\(_2\) production was not enhanced through the addition of carbon, suggesting that there are additional controlling mechanisms driving N\(_2\) production. This concurs with the lack of significant correlation between measured sedimentary TOC and N\(_2\) production. The lack of stimulation of N\(_2\) production after the addition of carbon has also been demonstrated in the oxygen minimum zone of the Arabian Sea, where denitrification (and anammox) was only enhanced at one out of 11 depths [36]. Instead, Bulow et al. [36] highlighted a correlation between denitrification and NO\(_2\) concentration, a factor which likely also plays a role in controlling denitrification in settlement pond systems but was not measured in

**Table 4.** The rate (nmol N cm\(^{-3}\) h\(^{-1}\)) of N\(_2\) production in three incubations (i.e. 24 h, 1.5 h and in the incubation with carbon additions).

<table>
<thead>
<tr>
<th>Pond</th>
<th>( \text{24 h incubation} )</th>
<th>( \text{1.5 h incubation} )</th>
<th>( \text{Carbon Incubation} )</th>
</tr>
</thead>
<tbody>
<tr>
<td>DNT</td>
<td>ANA</td>
<td>DNT</td>
<td>ANA</td>
</tr>
<tr>
<td>A</td>
<td>7.07(\pm)2.99 ND</td>
<td>7.97(\pm)3.35 ND</td>
<td></td>
</tr>
<tr>
<td>B</td>
<td>0.06(\pm)0.06 0.22(\pm)0.12</td>
<td>0.08(\pm)0.003 ND</td>
<td>0.004(\pm)0.003 0.03(\pm)0.02</td>
</tr>
<tr>
<td>C</td>
<td>0.04(\pm)0.003 ND</td>
<td>0.04(\pm)0.003 0.03(\pm)0.02</td>
<td></td>
</tr>
<tr>
<td>D</td>
<td>4.36(\pm)2.01 ND</td>
<td>6.32(\pm)4.16 0.48(\pm)0.48</td>
<td></td>
</tr>
</tbody>
</table>

DNT = denitrification; ANA = anammox.

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the present study. NO$_3^-$ concentration also regulates anammox activity in estuarine sediments [37], so future work should aim to correlate extractable NO$_3^-$, NO$_2^-$ and NH$_4^+$ with denitrification and anammox potentials to determine if these are driving process rates in settlement ponds.

It is also possible that the exogenous carbon source is instead stimulating nitrate ammonifiers (DNRA) and therefore competition for NO$_x$ as a substrate [38]. Of the added $^{15}$NO$_3^-$ only 7.9±2.7% was recovered as $^{15}$N$_2$, so a large portion (i.e. ~90%) of added $^{15}$NO$_3^-$ could be rapidly consumed by competing pathways such as DNRA or assimilation. The prevalence of DNRA or assimilation over denitrification will determine the balance between N being removed from the system through gaseous N$_2$ production, or conserved within the system [39–41]. Furthermore, although dominance of DNRA over denitrification and anammox has been demonstrated in tropical estuaries [42] and under fish cages [43], DNRA has never been quantified in tropical settlement ponds and warrants further investigation.

Another potential controlling factor may be the presence of free sulfides. Sulfur is cycled rapidly in tropical sediments [44], and is the most important anaerobic decomposition pathway in tropical benthic systems, occurring at rates of 0.2–13 mmol S m$^{-2}$ d$^{-1}$ and releasing free sulfides [45,46]. Free sulfides inhibit nitrification and therefore may be reducing N$_2$ production in the present study by reducing the amount of NO$_3^-$ available to denitrifiers [47]. Additionally, DNRA may be stimulated in the presence of sulfur,
increasing competition with denitrifiers for NO$_3^-$ [48]. Again, the effect of sulfur on N$_2$ production in tropical settlement ponds is largely unknown and further studies are needed to elucidate the potential of this factor on stifling N removal in settlement ponds.

**Denitrification verses anammox**

In our study denitrification was the dominant N$_2$ production pathway. In coastal, hyper-nutritited sediments, low N$_2$ production through anammox has been attributed to the limitation of NO$_2$ [49,50]. Further controlling factors for anammox are NH$_4^+$, total kilojoule nitrogen, TN, TP, salinity, redox state, and an inverse relationship with TOC [51]. Given these controlling factors anammox potential varies seasonally [51] and reported anammox contribution to N$_2$ production is highly variable with values of 1–8% [23], ≤3% [15], 10–15% [52], 19–33% [16], up to 65% [17], 2–67% [22] and 4–79% [53].

Anammox was detected in sediment collected in ponds B (24 h incubation), C (carbon incubation) and D (1.5 h incubation), notably, where overall N$_2$ production was exceptionally low. For example, during the 24 h incubation with sediment collected in pond B, N$_2$ production was lower than in sediment collected from all other ponds, but anammox contributed 95% to N$_2$ production. Low carbon oxidation rates and correspondingly low denitrification (and thus competition for substrate) have been proposed as the reason anammox contribution is high in environments where denitrification is low [17]. Bulow et al. [36] demonstrated that high anammox rates corresponded with low denitrification rates at one site in the oxygen minimum zone in the Arabian Sea. At this site both anammox and denitrification were stimulated by the addition of organic carbon. This suggests that N$_2$ production was carbon limited giving anammox the competitive advantage. In tropical estuary systems where high temperatures, low sediment organic content and low water column NO$_3^-$ concentrations prevail, the order of NO$_x$ reduction pathways is proposed to be DNRA$>\text{denitrification}>\text{anammox}$ [42].

The apparent detection of anammox in the presence of MeOH in sediments collected from Pond C is unusual given that anammox is inhibited by MeOH [28]. It is possible that during the 24 h incubation $^{15}$NH$_4^+$ was transformed through anoxic

**Table 5. The percent recovery of added $^{15}$N as labelled N$_2$ in three treatments.**

<table>
<thead>
<tr>
<th></th>
<th>$^{15}$N-NO$_3^-$</th>
<th>$^{15}$N-NH$_4^+$ &amp; $^{14}$N-NO$_3^-$</th>
<th>$^{15}$NH$_4^+$</th>
</tr>
</thead>
<tbody>
<tr>
<td>A</td>
<td>11.8±1.17</td>
<td>0.00±0.00</td>
<td>0.00±0.00</td>
</tr>
<tr>
<td>B</td>
<td>0.20±0.07</td>
<td>0.67±0.28</td>
<td>0.28±0.09</td>
</tr>
<tr>
<td>C</td>
<td>10.92±1.99</td>
<td>0.26±0.08</td>
<td>0.43±0.07</td>
</tr>
<tr>
<td>D</td>
<td>8.79±0.61</td>
<td>0.01±0.32</td>
<td>0.00±0.00</td>
</tr>
</tbody>
</table>

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**Figure 3.** Production of $^{29}$N$_2$ (black circles) and $^{30}$N$_2$ (white circles) over 24 h. $^{15}$N-N$_2$ production in the presence of $^{15}$N-NO$_3^-$ is represented in row 1, $^{15}$N-N$_2$ production in the presence of $^{15}$N-NH$_4^+$ and $^{14}$N-NO$_3^-$ is represented in row 2 and $^{15}$N-N$_2$ production in the presence of $^{15}$N-NH$_4^+$ is represented in row 3. Column 1 represents $^{15}$N-N$_2$ production in sediments collected from pond A, column 2 represents $^{15}$N-N$_2$ production in sediments collected from pond B, column 3 represents $^{15}$N-N$_2$ production in sediments collected from pond C and column 4 represents $^{15}$N-N$_2$ production sediments collected from pond D.

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nitrification [34], producing $^{15}$NO$_3^-$ and the resulting $^{15}$N$_2$ was produced as the result of denitrification.

**Settlement pond functioning and implications**

Microbial N$_2$ production has the potential to play a major role in removing N from aquaculture wastewater. However, we estimated that only 2.3% of total N added to the settlement pond via wastewater inputs and mineralization is removed through N$_2$ production. It is likely that the noxious compounds of H$_2$S and NH$_4^+$ are produced in settlement ponds when they are left unmanaged with no removal of settled particulate organic matter (sludge). These compounds have significant consequences for the inhibition of microbial processes that remove N from wastewater. In addition, H$_2$S accumulation causes a shift in the species of gaseous N produced from N$_2$ to N$_2$O due to the inhibition of the last step of denitrification [41]. This has detrimental consequences for global warming as N$_2$O is ~300 times more potent than CO$_2$ as a greenhouse gas whereas N$_2$ is relatively inert [55]. Future research should determine the concentration of H$_2$S at which the last reductive step of denitrification is inhibited and relate this to the amount of sludge that has built up in the settlement pond. We recommend that sludge be extracted at this point to prevent H$_2$S release and to prevent the recycling of soluble N through mineralization, DNRA or assimilation and subsequent senescence, as has been recommended for grow out ponds previously [30]. Innovative technology, such as anaerobic digesters and biogas capture, is required to convert the large volumes of sludge to a saleable product once removed from the pond. The simple management approach of removing sludge could have the added benefit of decreasing the incidence of competition between DNRA and denitrification thereby optimizing the denitrification and anammox processes for N$_2$ production. If N$_2$ production could be enhanced to the mean rate reported by Erler et al. [20] from a constructed wetland of 965 μmol N m$^{-2}$ d$^{-1}$, then 100% of total daily N inputs would be removed from settlement ponds every day. However, the estimates in the present study are based on a very simplistic understanding of the settlement pond functioning and the model requires better definition of the parameters. For example, accurate rates of NH$_4^+$ and DON production from the sediments are required to estimate N inputs accurately. Additionally, N$_2$ production was measured in the dry season in the present study when rates are likely lower than in the wet season. Wet season precipitation lowers the salinity in the ponds to 5% in some cases, which favors higher denitrification, lower DNRA and lower NH$_4^+$ fluxes [36]. Denitrification is further stimulated during periods of heavy precipitation due to increased NO$_3^-$ concentrations from land run-off [43]. An increased understanding of the temporal and spatial variability in N$_2$ production rates measured using intact core assays, instead of slurry assays, would also allow accurate predictions of N$_2$ production rates. Slurry assays only generate potential rates of N$_2$ production and we acknowledge that homogenizing sediments disrupts the sediment profile and can result in different nutrient availability than that which occurs in situ [57]. Additionally, an understanding of the rates of competing biogeochemical pathways such as DNRA and assimilation would enhance the accuracy of the model by including N retention rates into the model.

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**Author Contributions**

Conceived and designed the experiments: SC DE LT NP. Performed the experiments: SC DE. Analyzed the data: SC DE NP. Contributed reagents/materials/analysis tools: SC DE LT NP. Wrote the paper: SC DE LT RdN.

**References**


