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Running head: Industrial-scale aquaculture increases global warming

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Abstract

Fisheries capture has plateaued, creating ever-greater reliance on aquaculture to feed growing populations. Aquaculture volumes now exceed those of capture fisheries globally[1,2], with China dominating production through major land-use change; more than half of Chinese freshwater aquaculture systems having been converted from paddy fields[1,3]. However, the greenhouse gas (GHG) implications of this expansion have yet to be effectively quantified. Here we measure year-round methane (CH_4), nitrous oxide (N_2O) and carbon dioxide (CO_2) emissions from paddy fields and new, extensively managed crab aquaculture ponds. The conversion increased associated global warming potentials (GWP) from 8.15 ± 0.43 to 28.0 ± 4.1 Mg CO_2 eq ha^{-1}, primarily due to increased CH_4 emission. After compiling a worldwide database of different freshwater aquaculture systems, the top 21 producers were estimated to release 6.04 ± 1.17 Tg CH_4 and 36.7 ± 6.1 Gg N_2O in 2014. We found that 80.3% of total CH_4 emitted originated in shallow earthen aquaculture systems, with far lower emissions from intensified systems with continuous aeration[4]. We therefore propose greater adoption of aerated systems is urgently required to address globally significant rises in CH_4 emission from the conversion of paddy fields to aquaculture.

With increasing demand for animal proteins due to rising populations and a leveling off in capture fisheries, global aquaculture production has increased by 500% since the late-1980s, and now represents a major global industry[1]. In 2014, aquaculture volume amounted to 101 million tons (Mt) and is projected to reach 230 Mt by 2030, accounting for 62% of global fish and shellfish supply for human consumption[1,2]. This ever-expanding aquaculture sector relies heavily on application of aquafeeds[5,6] which increase nutrient loadings and carbon (C) burial in aquaculture systems and adjacent water bodies[7,8]. Only 25% (11–36%) of the nitrogen (N) consumed by fish was converted to biomass with the remainder excreted into water as un-ionized ammonia[9,10]. Likewise, a substantial proportion of feed C was transformed to CO_2 and CH_4 by animals and microbes[11] or buried in aquaculture systems[7]. In 2016, about 10.9 Tg C and 1.82 Tg N from the 39.9 Mt aquafeeds were estimated to be discharged to environments in global aquaculture[12]. Moreover, fertilizers are widely used in the extensive and semi-intensive aquaculture systems to stimulate phytoplankton production[13]. These intensive C and N loadings have the potential to drive aquaculture systems to become major anthropogenic sources of CH_4 and N_2O emissions.

Williams & Crutzen[14] tentatively estimated N_2O emission from the aquaculture sector at 0.09 Tg in 2008, accounting for 0.33% of global N_2O emission. Using the N_2O emission factor of influent N (EF_N = 1.80%) in wastewater treatment plants[15], global N_2O emission from aquaculture was estimated to increase from 0.15 Tg in 2009 to 0.60 Tg in 2030, which could contribute 5.72% of global anthropogenic N_2O emission[10]. However, large uncertainties in these estimates may arise from differences in management levels[16,17] and yield difference between species[17,18]. Besides N_2O, aquaculture ponds could be important anthropogenic CH_4 sources with characteristics of intensive C loading, shallow water and frequent mixing[19]. To date, >40% of worldwide aquaculture production has been carried out in earthen ponds, while estimates of overall CH_4 budgets in global aquaculture remain scarce.

China is the world's largest aquaculture producer, contributing ~60% of global volume[1]; furthermore the volume and area of that aquaculture is steadily rising[3]. Above 70% of Chinese freshwater aquaculture production is carried out in extensive and semi-intensive earthen ponds[3]. More and more paddy fields have been, and will continue to be, converted to aquaculture ponds. They currently account for 51.3% of Chinese inland fish ponds[3,18]. There is clearly an urgent need for greater appreciation of the costs associated with GHG emissions incurred during the ongoing unprecedented levels of conversion of paddy fields towards industrial-scale aquaculture.

Effect of conversion of paddy field to aquaculture on GHG emission

We measured year-round fluxes of CH_4, N_2O and CO_2 from three adjacent crab aquaculture ponds converted from paddy fields 12 years ago and neighboring paddy fields (PF) in the Tai Lake basin (31°02′N, 120°25′E; Supplementary Figs. 1 and 2) during 2013–2014. Wheat-rice rotation is the typical cropping system in this region. Urea was applied in PF at 150 and 280 kg N ha^{-1} during wheat and rice seasons, respectively. Crab ponds differed in size and water depth (Supplementary Table 1). They were not equipped with aerators but were fertilized during culturing. Chinese mitten crab (*Eriocheir sinensis*) were fed with commercial feed pellets, trash fish and corn seeds at the same rate in each pond during crab production period from March to October. Annual C and N inputs in crab ponds were 1.20 Mg C ha^{-1} and 244 kg N ha^{-1}, respectively (Supplementary Tables 2–4).

Annual CH_4 emission in PF was 218 ± 7.28 kg CH_4 ha^{-1} (Fig. 1a), which was located in the upper end of the previously reported ranges (98.3–240 kg CH_4 ha^{-1}) for paddy fields without organic amendment in this area[20]. However, conversion from PF to crab ponds sharply increased

CH_4 emission to 962 ± 62 kg CH_4 ha^{-1}; this value was higher than the summarized amount of 572 kg CH_4 ha^{-1} in permanently inundated temperate wetlands and the default emission factor (900 kg CH_4 ha^{-1}) for tropical inland freshwater wetlands proposed by the Intergovernmental Panel on Climate Change (IPCC; ref. 21).

The CH_4 EF_C of C inputs from feeds and fertilizer in crab ponds was estimated at up to 60.0% (Table 1), which may be attributed to the enhanced availability of labile organic substrates and highly anaerobic environment in crab ponds. The C output as harvested crab was 0.19 Mg C ha^{-1} (Supplementary Table 4), accounting for 16.1% of the C inputs excluding the photosynthates by submerged macrophytes. The remaining 1.04 Mg C ha^{-1} was deposited into sediments as unconsumed feed and feces, which led mean dissolved organic C (DOC) concentrations in pond sediments to reach 7.97-fold greater than that of PF (Supplementary Table 1). Additionally, organic compounds in feed remnants and feces such as starch and protein can be more easily decomposed[22] to methanogenic substrates than crop residues in PF. Moreover, pond sediments were permanently inundated, thereby creating anaerobic environments ideal for methanogenesis.

Annual N_2O emission in PF was 7.11 ± 0.23 kg N_2O ha^{-1} (Fig. 1b). The EF_N of fertilizer-N applied was 1.05%, closing to the IPCC default value (1.00%) for agricultural soils[23]. Conversion from PF to crab ponds significantly decreased annual N_2O emission by 95.4% to 0.33 ± 0.07 kg N_2O ha^{-1}, with an EF_N of 0.09 ± 0.02% (Table 1). The lower N_2O emission in crab ponds was disproportionate to the differences in N application rates (244 vs 430 kg N ha^{-1}), let alone the relatively higher total inorganic N content in pond sediments (Supplementary Tables 1). Nitrous oxide is derived from both nitrification and denitrification, although denitrification produces more N_2O (ref. 24). It is likely that the much lower redox potential (–124 to –160 mV) suppressed nitrification in pond sediments, which reduced overall NO_3^- concentrations to <1 mg N kg^{-1}. This concentration was lower than the threshold value of 5 mg N kg^{-1} for active denitrification[25]. Moreover, the high DOC concentrations and anaerobic conditions permit N_2O to be further reduced to N_2 through denitrification[26].

Using the net ecosystem C balance method, annual loss of soil organic C (SOC) in PF was estimated to be 0.04 ± 0.05 Mg C ha^{-1} (Table 1), which fell in the range of –0.27 to 0.67 Mg C ha^{-1} estimated previously in paddy fields of Tai Lake basin[27]. The CO_2 fluxes measured by transparent chambers in crab ponds were regarded as net ecosystem exchange. On an annual basis, crab ponds were weak net CO_2 sources, releasing 0.13–1.99 Mg CO_2 ha^{-1} (Fig. 1c).

Conversion from PF to extensive crab ponds increased the 100-yr GWP from 8.15 ± 0.43 to 28.0 ± 4.1 Mg CO_2 eq ha^{-1}, mainly due to increased CH_4 emission with a contribution of 96.3% (Table 1). Our results contrast with those of Liu et al.[18], who reported such conversion significantly reduced CH_4 and N_2O emissions by 48% and 56%, respectively. Annual CH_4 emission in Liu's ponds (equipped with aerators and classified as semi-intensive, see below) was just 32.6 kg CH_4 ha^{-1} despite the much greater feeding rate and higher sediment DOC concentration compared to test extensive ponds. Hence, substrate availability was not the limiting factor for CH_4 emissions in feeding aquaculture systems, while oxygen exposure by aeration was the key factor affecting CH_4 emissions. Our results highlight that GHG emissions clearly differ from one aquaculture system to another, greatly depending on the intensity of operational management. This observation illustrates the potential for mitigating the effects of future paddy field conversion through careful management.

Global CH_4 and N_2O budgets of freshwater aquaculture

Here, we classified aquaculture into four systems: rice-fish, extensive, semi-intensive and intensive based on local conditions and aquaculture facilities especially whether aerators are used or not (see Methods). We compiled a worldwide database of CH_4 and/or N_2O emissions that were measured in 45 inland freshwater aquaculture systems during 2003–2015 (Supplementary Table 5). Land-use and production statistics were also compiled for different aquaculture systems of top 21 freshwater aquaculture producers (Supplementary Table 6); however, data from extensive and semi-intensive systems were pooled because of unavailability of aerator-use data for separate classification. In 2014, the top 21 producers contributed 97.5% of global freshwater aquaculture volume[1].

The synthesized data show that CH_4 fluxes ranged from –0.03 to 37.0 mg CH_4 m^{-2} h^{-1} in rice-fish, extensive, and semi-intensive systems. Mean CH_4 flux in rice-fish system was the highest at 12.6 ± 3.9 mg CH_4 m^{-2} h^{-1}, followed by extensive and semi-intensive systems (Fig. 2a). The absence of CH_4 flux in intensive systems can be attributed to a combination of continuous aeration, water exchange and a lack of habitats for methanogens[4]. The rice-fish system also had the highest mean N_2O flux followed by semi-intensive and extensive systems (28.4 ± 9.8 and 7.56 ± 3.02 μg N_2O m^{-2} h^{-1}, respectively; Fig. 2b). The EF_N and yield-scale N_2O EF (EF_Y) in intensive system were 1.16 ± 0.18% and 2.48 ± 0.42 g N_2O kg^{-1} yield, respectively, which were significantly higher than the corresponding values in extensive (0.24 ± 0.10% and 0.66 ± 0.22 g N_2O kg^{-1} yield) and semi-intensive (0.35 ± 0.16% and 0.88 ± 0.41 g N_2O kg^{-1} yield) systems. The EF_Y in intensive systems was close to the IPCC default EF_Y (2.66 g N_2O kg^{-1} yield) that is widely used in model estimate for aquaculture[10,21], but was 2.75- and 1.82-fold greater than that for extensive and semi-intensive systems, respectively. Considering the large volume of extensive and semi-intensive aquaculture (Supplementary Table 6), previous estimates[10,14] of global aquaculture N_2O emission may have been overestimated because of the higher default EF_Y mentioned above.

The estimated CH_4 and N_2O emissions from the top 21 producers in 2014 were 6.04 ± 1.17 Tg

CH_4 and 36.7 ± 6.1 Gg N_2O, respectively (Table 2), which accounted for 1.82% and 0.34% of global anthropogenic CH_4 and N_2O emissions, respectively. Methane was a key contributor (94.6%; Fig. 2e) to GWP in freshwater aquaculture, of which 1.19 ± 0.27 Tg CH_4 was emitted from rice-fish system and 4.85 ± 1.04 Tg CH_4 from extensive plus semi-intensive systems. To our knowledge, this is the first global estimate of CH_4 emission from freshwater aquaculture. Our estimated total N_2O emission was much lower than the previous estimates of 90 Gg N_2O (ref. 14) and 146 Gg N_2O (ref. 10) of global aquaculture. Extensive plus semi-intensive systems contributed 87.0% of global volume of freshwater aquaculture, meanwhile, were the largest CH_4 and N_2O emitter (80.3% and 45.2%, respectively) from this sector. Intensive systems accounted for 8.89% of the production, 27.0% of total N_2O emissions but negligible CH_4 emissions. Rice-fish systems represented only 4.30% of aquaculture volume, yet they accounted for 19.7% and 27.8% of CH_4 and N_2O budgets, respectively.

The greenhouse gas intensity (GHGI, GWP/yield) was 3.59 ± 0.74 kg CO_2 eq kg^{-1} yield in extensive plus semi-intensive systems, which was 4.46-fold greater than that in intensive systems (0.66 ± 0.11 kg CO_2 eq kg^{-1} yield; Fig. 2f). Therefore, if half of the current productions from extensive plus semi-intensive systems (19.5 Mt) are replaced by intensive systems, the GWP of CH_4 and N_2O emissions from freshwater aquaculture (excluding rice-fish) will be reduced by 40.1% from 143 Tg CO_2 eq to 85.6 Tg CO_2 eq.

China has emerged as the world's largest freshwater aquaculture emitter of CH_4 (4.10 ± 0.10 Tg yr^{-1}) and N_2O (22.8 ± 7.1 Gg yr^{-1}), contributing 68.0% and 62.1% of global budgets from the sector, respectively. In China, CH_4 emissions from freshwater aquaculture with 7.57×10^6 ha equates to 36.5% of total CH_4 emissions from paddy fields, natural wetlands and lakes (11.3 Tg CH_4 yr^{-1}; ref. 28). Since 83.0% of Chinese freshwater aquaculture CH_4 emissions originate from extensive plus semi-intensive systems, a substantial reduction in emissions could be achieved through improved management practices, such as installing more efficient aerators in earthen ponds and implementing optimized feeding strategies for reducing feed waste.

In conclusion, the conversion of paddy fields to extensive crab aquaculture ponds sharply increased GWP, primarily through a drastic increase in CH_4 release. Our findings emphasize the need to assess the climatic impacts of land-use shifts towards industrial-scale aquaculture. Methane is the most important GHG in freshwater aquaculture compared with N_2O, and it was primarily sourced from extensive plus semi-intensive systems. Our findings indicate that effective management of extensive and semi-intensive systems through conversion to intensive systems is urgently required to mitigate GHG emissions from the unprecedented growth of aquaculture.

Methods

Methods, including statements of data availability and any associated accession codes and references, are available in the online version of this paper.

Data availability

The authors declare that the data supporting the findings of this study are available within the article and its supplementary information files.

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Additional information

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Author contributions

W.D., J.Y., and D.L. designed the study; J.X. led the GHG fluxes and auxiliary measurements with the support of T.H., S.K., and Y.L. Site selection and set-up was carried out by J.Y., and D.L. H.K. and C.F. were the key international collaborators during this research. The manuscript was drafted by J.Y., H.K., W.D. and C.F with all authors contributing to the final version.

Competing financial interests

The authors declare no competing financial interests.

Methods

Site description Field experiments were carried out in a conventional paddy field (PF) and three adjacent crab ponds in the Tai Lake basin (31°02′N, 120°25′E), Suzhou City, Jiangsu Provence, China (Supplementary Fig. 1). This region is characterized by a subtropical monsoon climate with the long-term (1981–2010) mean annual air temperature of 16.5°C and precipitation of 1176 mm (http://cdc.nmic.cn/home.do). Paddy fields accounted for 65% of total cropland in this region, however, they are being rapidly converted to aquaculture ponds due to the greater economic benefits from the latter since 1980s (ref. 31).

Soil was developed from alluvial sediments of the Yangtze River, and classified as stagnic Anthrosols based on the USDA soil taxonomy. The surface soil (0–20 cm) had a pH (H_2O) of 5.95, bulk density of 1.25 g cm^{-3} and a loam texture with 40% sand, 34% silt and 26% clay, and contained 20.3 g kg^{-1} organic C and 1.81 g N kg^{-1} total N. Three neighboring aquaculture ponds for Chinese mitten crab (*Eriocheir sinensis*) cultivation were converted from paddy fields in 2001.

Experimental design and field management Four independent 3 × 8 m^2 plots were established in PF in November 2012. Winter wheat (*Triticum aestivum* L., Yangfumai 4) and summer rice (*Oryza sativa* L., Wuyunjing 23) was rotated during the period from November 2012 to May 2014. During wheat season, urea was applied at the rate of 150 kg N ha^{-1}, with basal and supplemental fertilization ratio of 40%:60%. During rice season, urea was applied at the rate of 280 kg N ha^{-1}, with the basal and supplemental fertilizer ratio of 50%:50%. Calcium superphosphate (40 and 125 kg P_2O_5 ha^{-1} for wheat and rice, respectively) and potassium sulfate (60 and 125 kg K_2O ha^{-1} for wheat and rice, respectively) were applied as basal fertilizers (Supplementary Table 2). The row distance was 25 cm for rice and wheat, and the hill distance was 15 cm for rice. No irrigation was performed during wheat season, while rice was managed under a typical water regime mode of flooding-midseason drainage-reflooding-moist irrigation (F-D-F-M). Crop grain and straw were harvested and oven dried at 60°C until a constant weight.

Parallel field experiments were conducted in three neighboring crab ponds with different size (CP1, 1.71 ha; CP2, 0.71 ha; CP3, 0.09 ha), from March 2013 to March 2014. Monoculture of Chinese mitten crab was employed at the same stocking density of 15000 ind ha^{-1} for each pond. The submerged western waterweed (*Elodea nuttallii*) naturally vegetated in ponds and provided molting shelters and foods for crabs. Feeds and fertilizers were applied at the same rates in each pond. Snails (*Bellamya quadrata*) were introduced into the ponds twice at the rates of 600 and 400 kg ha^{-1} on April 4 and June 20, respectively, to filter feed residue and provide supplementary foods for crabs (Supplementary Tables 2 and 3). Crabs were fed with commercial feed pellets (2050 kg ha^{-1}) (Purina Co. Ltd., Jiaxing, China), trash fish (1250 kg ha^{-1}) and corn seeds (1150 kg ha^{-1}), twice per day on 9:00 a.m. and 17:00 p.m. until the crabs were harvested. In order to stimulate phytoplankton and waterweed production, cake manure (residue of de-oiled oil seeds) at 40 kg ha^{-1} was applied as basal fertilizer while urea, compound fertilizer and calcium superphosphate were applied at the rate of 130, 100 and 200 kg ha^{-1}, respectively, with four splits of 25%:25%:25%:25% on March 29, June 5, August 12 and September 27. Annual inputs of C and N to crab ponds were 1.20 Mg C ha^{-1} and 244 kg N ha^{-1}, respectively. Water was constantly maintained all-year round, while the water depth differed between ponds. Crab harvest started from 1 to 30 October 2013, depending on crab maturity. Crab yield was expressed as fresh weight (Supplementary Table 4). Details management practices in the two systems are shown in Supplementary Table 3.

Measurement of GHG fluxes Wooden boardwalks were installed in each plot to facilitate collecting gas samples and measuring the auxiliary parameters (Supplementary Fig. 2). The static closed chamber technique was used to measure GHG fluxes; in PF, PVC chamber collars (50 cm × 50 cm × 20 cm) with a water-filled channel were inserted into the soil at a depth of 15 cm. In crab ponds, a specially designed system, which included four stainless steel pegs for fixing the system and two adjustable crossbars for elevating or lowering the chamber collars with the fluctuation of water level, were installed along the boardwalks to minimize water wave impact on gas sampling. Three PVC chamber collars were placed on the crossbars in each pond. If necessary, the crossbars together with the chamber collars were adjusted to the best position one day before sampling. The transparent Plexiglass chambers (50 cm × 50 cm × 15 cm) in crab ponds and the stainless steel chambers (50 cm × 50 cm × 50 cm) insulated with white foam in PF were used. See Yuan et al.[32] for further detailed information of the devices.

The GHG fluxes were measured twice weekly in crab ponds during crab production period from March to October and weekly during period without crab production from November to February (Supplementary Fig. 3). In PF, GHG fluxes were measured twice weekly from April to November and weekly from December to March. Gas sampling was conducted at 08:00–10:00 local time to minimize diurnal variation in the flux pattern. During sampling, the chamber was fitted into the water trough of the chamber collars. Each time, four gas samples of the chamber headspace were drawn using a 50-mL syringes at 0, 10, 20, and 30 min after closure and injected into 22-mL pre-evacuated glass vials. Air temperature inside the chamber was simultaneously measured with a mercury thermometer. Concentrations of CH_4, N_2O and CO_2 were determined by a gas chromatograph (Agilent 7890, Santa Clara, CA, USA) equipped with a flame ionization detector for CO_2 and CH_4 and a ^{63}Ni electron capture detector for N_2O. The gas standards were provided by the National Research Center for Certified Reference Materials, Beijing, China. The precision for GHG concentrations was ±0.5% based on repeated measurements of gas standards. The GHG fluxes were calculated using a linear least squares fit to the four points in the time series of concentration for each plot. Data were omitted if the slope of the linear fitting had $R^2 < 0.90$. Since the opaque chambers were used in PF, the measured CO_2 fluxes were ecosystem respiration (*Re*); in contrast, CO_2 fluxes in crab ponds measured by transparent chambers were net ecosystem exchange[32]. The dataset of GHG fluxes were supplied as Supplementary Table 7.

Annual or seasonal cumulative CH_4 (kg CH_4 ha^{-1}), N_2O (kg N_2O ha^{-1}) and CO_2 (kg CO_2 ha^{-1}) emissions (E) were calculated using the following equation:

$$E = \sum_{i=1}^{n} (f_i + f_{i+1})/2 \times (t_{i+1} - t_i) \times 24 \times 10^{-2}$$

where f represents the flux of CH_4 (mg CH_4 m^{-2} h^{-1}) or N_2O (mg N_2O m^{-2} h^{-1}) or CO_2 (mg CO_2 m^{-2} h^{-1}); i is the ith measurement; $(t_{i+1} - t_i)$ is the days between two adjacent measurements; and 24×10^{-2} was used for unit conversion.

Auxiliary measurements Redox potential of the intact soil and sediment at 10 cm depth was measured *in situ* using a PHB-6 pH/mV meter (Jiaoyuan Instrument, Yancheng, China). The soil of PF or sediment of ponds at 10 cm depth was collected weekly using a Russian corer for mineral N and dissolved organic C measuring. The NH_4^+ and NO_3^- were extracted with 2 M KCl solution (shaken for 1 h and then filtered); extracts were filtered and analyzed on a continuous-flow analyzer (SAN++, Skalar, Breda, the Netherlands). Dissolved organic C was extracted with deionized water (shaken for 30 min at 25°C, centrifuged for 25 min at 4000 rpm, and filtered through 0.45-µm membrane filter) and measured on a TOC analyzer (TOC Vcph, Shimadzu, Kyoto, Japan). Soil organic C (SOC) and total N contents were determined by the wet-oxidation redox method and the

Kjeldahl procedure, respectively[33].

Estimates of SOC change in paddy field and GWP The SOC change (δSOC) in PF was estimated from the net ecosystem C balance (NECB) using a coefficient of 0.213 for paddy soils in this study[29], namely, the conversion rate of organic C gain to SOC is 213 g C kg^{-1}. The NECB of the short-plant croplands was calculated according to Ma et al.[27]:

$$\mathrm{NECB} = \mathrm{GPP} - \mathrm{Re} - \mathrm{Harvest} - \mathrm{CH_4} + \mathrm{Manure}$$

where GPP (gross primary production) is inferred from NPP (net primary production) via the NPP/GPP ratio of 0.58 in this region deduced by Zhang et al.[34]; *Re*, CH_4 and manure are the C exchange through ecosystem respiration, CH_4 emission, and manure application, respectively; Harvest is the C of removed straw and grain, which was calculated based on biomass yields, and C and N contents in straw and grain (Supplementary Table 4). The NPP includes net primary productions of grain, straw, root, litter and rhizodeposit, according to Ma et al.[27].

The GWP (Mg CO_2eq ha^{-1}) in PF is calculated by the following equation[35]:

$$\mathrm{GWP} = 28 \times \mathrm{CH_4} + 265 \times \mathrm{N_2O} - 44/12 \times \delta\mathrm{SOC}$$

and for crab ponds:

$$\mathrm{GWP} = 28 \times \mathrm{CH_4} + 265 \times \mathrm{N_2O} + 1 \times \mathrm{CO_2}$$

where CH_4, N_2O and CO_2 denote annual emissions of CH_4 (Mg CH_4 ha^{-1}), N_2O (Mg N_2O ha^{-1}) and CO_2 (Mg CO_2 ha^{-1}), respectively.

Data collection and classification of global freshwater aquaculture As mentioned above, there are large uncertainties in previous model estimates of global aquaculture N_2O emissions by using EFs of applied N and fish yields: First, the N_2O EFs are highly dependent on management levels in the aquaculture system. For example, yield-scale N_2O EF (EF_Y) of carp was 1.07 g N_2O kg^{-1} yield in an intensive rearing system[16] but was only 0.28 g N_2O kg^{-1} yield in a semi-intensive earthen pond[17]; Secondly, the EF_Y can be biased by the major yield difference between species. For instance, although the direct N_2O emission rates in two adjacent semi-intensive aquaculture ponds were comparable, the EF_Y measured in crab ponds was 8.11-fold greater than that in carp ponds due to the magnitude difference in yields[17,18].

Here, we compiled a worldwide database of GHG emissions measured in the inland freshwater aquaculture systems (Supplementary Table 5). We identified potential published studies for inclusion in the database using Web of Science with the keywords 'greenhouse gases or CH_4 or N_2O' and 'aquaculture or fish farming or rice fish or aquaponics'. Twenty-four studies fell within the inland freshwater aquaculture and met the following criteria: (i) field measurement of CH_4 and/or N_2O emissions was carried out on a per hectare or per fish yield basis; (ii) type of aquaculture system with or without aerator use was reported; (iii) the N input and yield in intensive systems were listed (see below). The dataset include 45 CH_4 and/or N_2O emission measurements across 19 sites between 2003 and 2015.

Generally, the aquaculture systems are classified based on production per unit volume or per unit area[36]; however, when estimating the regional or global GHG emissions, such classification might be unfit due to lack of the available production data counted by volume or area and deficiency of the cross-species classification criteria for big differences in production performance between culture species. Here, we classified four systems: rice-fish, extensive, semi-intensive and intensive based on the local conditions and aquaculture facilities especially aerator use or not. Actually, the stocking density and production are associated with investment on infrastructure especially aeration equipment[36], because the dissolved oxygen in fish ponds should be maintained >5.0 mg L^{-1}, theoretically[37].

• Rice-fish systems include integrated rice field or rice field-pond complex and are used to produce fish and other aquatic animals.

• Extensive aquaculture systems involve excavated earthen ponds, irrigation canals and ditches, small lakes and reservoirs used for fish farming. Extensive systems have low stocking density, with natural productivity or limited supplemental feeds and no aerator system.

• Semi-intensive aquaculture systems include excavated earthen ponds, irrigation canals and ditches, small lakes and reservoirs, have higher stocking densities than extensive systems, and are equipped with aerators and managed with artificial feeds and intermittent aeration.

• Intensive aquaculture systems, which utilize man-made rearing units such as concrete/canvas tanks, raceways recirculating systems, have high stocking rates and complete diet management, intensive and continuous aeration, and frequent or continuous water exchange. The cage and pen culture performed in open water bodies like rivers, lakes and reservoirs are also classified as intensive aquaculture because of the high stocking rates and sufficient dissolved oxygen supply from the constant water exchange.

Global inventory of the land use and production statistics are also compiled in different aquaculture systems of the major freshwater aquaculture producers (Supplementary Table 6), however, data of extensive and semi-intensive systems were pooled because of lack of aerator use data to classify each other. Data were derived from the official fisheries statistics for 2014. In case 2014 data were not available, the most recent data were used. If the national official statistical data were not available, the FAO estimate (National Aquaculture Sector Overview) or private survey data were used. Further details on the statistics used are provided in the Supplementary materials.

Estimation of global CH_4 and N_2O budgets We estimated N_2O emissions from intensive systems by multiplying EF_Y by production. Methane emission from intensive systems was recognized as negligible because the aerobic condition limited CH_4 production in such systems[4]. While CH_4 and N_2O emissions from rice-fish, extensive, and semi-intensive aquaculture systems were estimated by multiplying mean emission rates by area, because (i) the yield EF for CH_4 was generally unavailable in literature; and (ii) the EF_Y would be biased by the huge yield difference between species in extensive and semi-intensive systems. Additionally, when estimating CH_4 emission from rice-fish systems, the CH_4 fluxes (32–37 mg CH_4 m^{-2} h^{-1}) measured in Bangladesh[30] were excluded from mean emission rates, because of the extremely high emission rates and relatively small area of rice-fish in Bangladesh (~3.97% of global rice-fish area).

It should be noted that our preliminary estimates possess some uncertainties. First, field measurements of CH_4 and N_2O fluxes were mainly conducted during the feeding period, may result in overestimation of CH_4 and N_2O emissions; secondly, only averaged CH_4 and N_2O fluxes in extensive and semi-intensive systems were set up due to the absence of detailed aquaculture facilities data; thirdly, there was no detailed information relative to land use and production in aquaculture in many main producers (e.g. Brazil, Nigeria). More field measurements along with detailed national aquaculture information in those countries are required to obtain more reliable estimates. Moreover, our estimates only focused on the direct CH_4 and N_2O emissions, however, GHG emission from adjacent water bodies can also be enhanced by the nutrients loading caused by water exchange in some aquaculture systems (especially intensive systems). Hence, these potential indirect emissions should be considered in future estimates.

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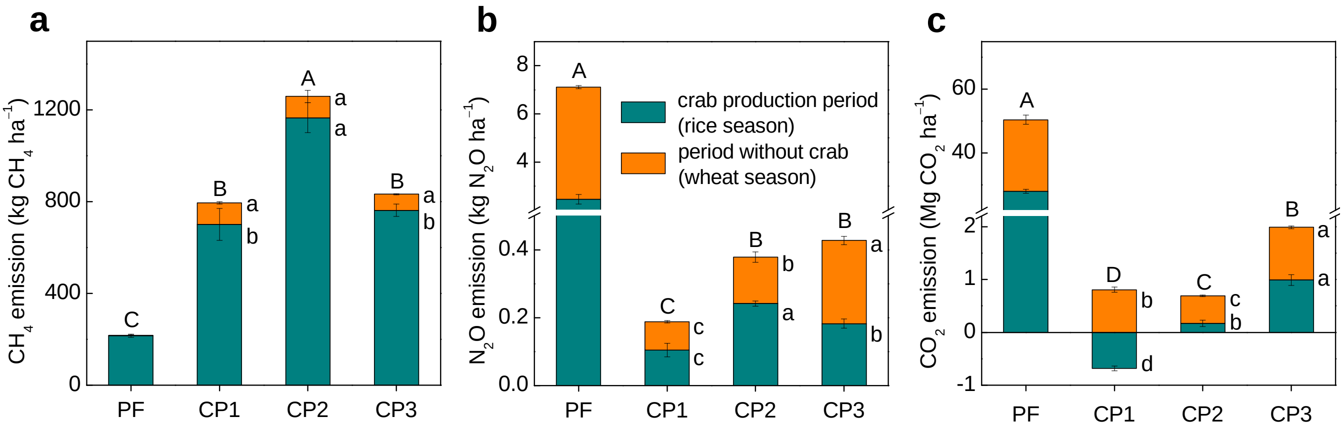
Figure legends:

Figure 1. Annual CH_4, N_2O and CO_2 emissions from the paddy field (PF) and crab ponds (CP) during 2013–2014. **a**, CH_4, **b**, N_2O, **c**, CO_2. Vertical bars represent standard errors of the means (*n* = 4 for PF and *n* = 3 for crab ponds). Three crab ponds had different size and water depth (Supplementary Table 1). 'A', 'B', and 'C' denote significant differences between sites ($P < 0.05$, ANOVA, Tukey's HSD test) during the entire year; 'a', 'b', and 'c' denote significant differences between crab ponds during the crab production period or during the period without crab production. CO_2 release from PF was calculated from soil organic C change estimates using the net ecosystem C balance method.

Figure 2. Literature-sourced greenhouse gas emission factors of different aquaculture. **a**, mean CH_4 emission rate, **b**, mean N_2O emission rate, **c**, N_2O emission factor of applied N (EF_N), **d**, yield based N_2O emission (EF_Y). Boundaries of the boxes indicate the first and third quartiles, line within the box and the white square represent the median and average, respectively. Whiskers mark the 10th and 90th percentiles, and the outliers are shown as dots. **e**, global warming potential (GWP), **f**, greenhouse gas intensity (GHGI, GWP/yield). Vertical bars represent standard errors of the means. Aquaculture systems are classified based on the local conditions and aquaculture facilities especially whether aerators were used or not.



a
b
c
CH_4 emission (kg CH_4 ha^{-1})
N_2O emission (kg N_2O ha^{-1})
CO_2 emission (Mg CO_2 ha^{-1})
crab production period
(rice season)
period without crab
(wheat season)
PF
CP1
CP2
CP3

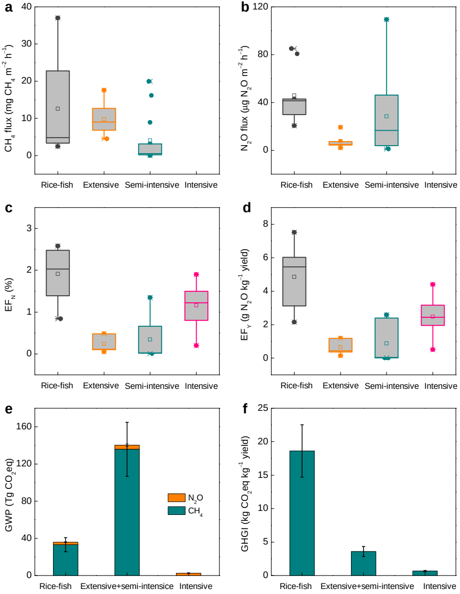

a
b
c
d
e
f
CH4 flux (mg CH4 m-2 h-1)
N2O flux (μg N2O m-2 h-1)
EFN (%)
EFY (g N2O kg-1 yield)
GWP (Tg CO2eq)
GHGI (kg CO2eq kg-1 yield)
Rice-fish
Extensive
Semi-intensive
Intensive
Extensive+semi-intensice
Extensive+semi-intensive
N2O
CH4

Table 1 Annual GHG emissions, net GWP and emission factors of CH_4 and N_2O in paddy field and crab ponds

Systems	CH_4 (kg CH_4 ha^{-1})	N_2O (kg N_2O ha^{-1})	CO_2* (Mg CO_2 ha^{-1})	C input† (Mg C ha^{-1})	N input† (kg N ha^{-1})	δSOC‡ (Mg C ha^{-1})	Net GWP§ (Mg CO_2eq ha^{-1})	EF_C†† (%)	EF_N¶ (%)	EF_Y¶ (g N_2O kg^{-1} yield)
Paddy field	218 ± 7b	7.11 ± 0.23a	50.6 ± 0.9a	–	430	–0.04 ± 0.05	8.15 ± 0.43b	–	1.05 ± 0.03a	0.56 ± 0.02a
Crab ponds	962 ± 149a	0.33 ± 0.07b	0.93 ± 0.55b	1.20	244	–	28.0 ± 4.1a	60.0 ± 9.3	0.09 ± 0.02b	0.30 ± 0.07b

* The value is ecosystem respiration in paddy field and net ecosystem CO_2 exchange for crab ponds. † Calculated by application rates and C and N contents of the fertilizers and feeds (see Supplementary Tables 2–4). ‡ Estimated from the net ecosystem carbon balance (NECB) using a coefficient of 0.213 for paddy soils[29]. § Net GWP = $28\times CH_4+265\times N_2O-44/12\times\delta SOC$ for paddy field, and net GWP = $28\times CH_4+265\times N_2O+1\times CO_2$ for crab ponds. †† The direct emission factor of C for CH_4 (EF_C) is calculated by dividing annual CH_4 emission by total C input[21]. ¶ The direct emission factor of N for N_2O (EF_N) and yield-scaled emission factor for N_2O (EF_Y) are calculated by dividing annual N_2O emission by total N input and grain/crab yield, respectively. Values are means ± standard errors.

Table 2 Direct CH_4 (Gg CH_4 yr^{-1}) and N_2O (Mg N_2O yr^{-1}) emissions from different freshwater aquaculture systems in global top 21 producers in 2014

Country/region	Rice-fish systems*		Extensive plus semi-intensive systems*		Intensive systems†	Total‡	
	CH_4	N_2O	CH_4	N_2O	N_2O	CH_4	N_2O
China	696	5,988	3,408	11,653	5,152	3,524	22,793
India	108	925	487	1,667	–	512	2,591
Indonesia	66	571	91	313	1,955	142	2,839
Vietnam	19	161	173	590	344	162	1,095
Bangladesh	–	–	323	1,106	4	268	1,109
Myanmar	–	–	50	172	0	42	172
Brazil	–	–	45	153	430	37	584
Thailand	2	15	71	244	91	61	350
Nigeria‡	–	–	–	–	–	–	–
Philippines	–	–	8	28	373	7	401
Iran	0	2	28	97	316	24	415
USA	15	128	35	119	76	44	323
Egypt	268	2,306	1	3	442	269	2,752
Pakistan	–	–	8	28	0	7	28
Taiwan Province of China	0	0	34	116	0	28	116
Russia	–	–	57	194	71	47	265
Cambodia	0	2	1	3	208	1	213

Uganda	–	–	6	19	67	5	86
Lao PDR	2	20	21	71	55	20	146
Turkey	–	–	0	0	268	0	268
Malaysia	12	101	3	12	50	15	162
Top 21 subtotal	1,188	10,219	4,851	16,586	9,903	6,039	36,709

* Calculated by mean CH_4 and N_2O emission rates (Fig. 2) and the area for aquaculture (Supplementary Table 6) collected from the literature. Rates of CH_4 emission from rice-fish system in Bangladesh were excluded when calculating[30]. † Calculated by averaged yield-scaled emission factor for N_2O (EF_Y) (Fig. 2d) and volume of production from intensive aquaculture. The direct emission rate of CH_4 from intensive system was estimated at 0 according to Hu et al.[4]. ‡ No official or private statistics is available about area and production from different systems in Nigeria.