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1 **Rapid growth in greenhouse gas emissions from the adoption of industrial-scale aquaculture**

2

3 **Running head:** Industrial-scale aquaculture increases global warming

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17

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23 **Abstract**

24 Fisheries capture has plateaued, creating ever-greater reliance on aquaculture to feed growing
25 populations. Aquaculture volumes now exceed those of capture fisheries globally^{1,2}, with China
26 dominating production through major land-use change; more than half of Chinese freshwater
27 aquaculture systems having been converted from paddy fields^{1,3}. However, the greenhouse gas
28 (GHG) implications of this expansion have yet to be effectively quantified. Here we measure
29 year-round methane (CH₄), nitrous oxide (N₂O) and carbon dioxide (CO₂) emissions from paddy
30 fields and new, extensively managed crab aquaculture ponds. The conversion increased associated
31 global warming potentials (GWP) from 8.15 ± 0.43 to 28.0 ± 4.1 Mg CO₂ eq ha⁻¹, primarily due to
32 increased CH₄ emission. After compiling a worldwide database of different freshwater aquaculture
33 systems, the top 21 producers were estimated to release 6.04 ± 1.17 Tg CH₄ and 36.7 ± 6.1 Gg N₂O
34 in 2014. We found that 80.3% of total CH₄ emitted originated in shallow earthen aquaculture
35 systems, with far lower emissions from intensified systems with continuous aeration⁴. We therefore
36 propose greater adoption of aerated systems is urgently required to address globally significant rises
37 in CH₄ emission from the conversion of paddy fields to aquaculture.

38 With increasing demand for animal proteins due to rising populations and a leveling off in capture
39 fisheries, global aquaculture production has increased by 500% since the late-1980s, and now
40 represents a major global industry¹. In 2014, aquaculture volume amounted to 101 million tons (Mt)
41 and is projected to reach 230 Mt by 2030, accounting for 62% of global fish and shellfish supply for
42 human consumption^{1,2}. This ever-expanding aquaculture sector relies heavily on application of
43 aquafeeds^{5,6} which increase nutrient loadings and carbon (C) burial in aquaculture systems and
44 adjacent water bodies^{7,8}. Only 25% (11–36%) of the nitrogen (N) consumed by fish was converted
45 to biomass with the remainder excreted into water as un-ionized ammonia^{9,10}. Likewise, a
46 substantial proportion of feed C was transformed to CO₂ and CH₄ by animals and microbes¹¹ or
47 buried in aquaculture systems⁷. In 2016, about 10.9 Tg C and 1.82 Tg N from the 39.9 Mt aquafeeds
48 were estimated to be discharged to environments in global aquaculture¹². Moreover, fertilizers are
49 widely used in the extensive and semi-intensive aquaculture systems to stimulate phytoplankton
50 production¹³. These intensive C and N loadings have the potential to drive aquaculture systems to
51 become major anthropogenic sources of CH₄ and N₂O emissions.

52 Williams & Crutzen¹⁴ tentatively estimated N₂O emission from the aquaculture sector at 0.09 Tg
53 in 2008, accounting for 0.33% of global N₂O emission. Using the N₂O emission factor of influent N
54 (EF_N = 1.80%) in wastewater treatment plants¹⁵, global N₂O emission from aquaculture was
55 estimated to increase from 0.15 Tg in 2009 to 0.60 Tg in 2030, which could contribute 5.72% of
56 global anthropogenic N₂O emission¹⁰. However, large uncertainties in these estimates may arise
57 from differences in management levels^{16,17} and yield difference between species^{17,18}. Besides N₂O,
58 aquaculture ponds could be important anthropogenic CH₄ sources with characteristics of intensive C
59 loading, shallow water and frequent mixing¹⁹. To date, >40% of worldwide aquaculture production
60 has been carried out in earthen ponds, while estimates of overall CH₄ budgets in global aquaculture

61 remain scarce.

62 China is the world's largest aquaculture producer, contributing ~60% of global volume¹;
63 furthermore the volume and area of that aquaculture is steadily rising³. Above 70% of Chinese
64 freshwater aquaculture production is carried out in extensive and semi-intensive earthen ponds³.
65 More and more paddy fields have been, and will continue to be, converted to aquaculture ponds.
66 They currently account for 51.3% of Chinese inland fish ponds^{3,18}. There is clearly an urgent need
67 for greater appreciation of the costs associated with GHG emissions incurred during the ongoing
68 unprecedented levels of conversion of paddy fields towards industrial-scale aquaculture.

69

70 **Effect of conversion of paddy field to aquaculture on GHG emission**

71 We measured year-round fluxes of CH₄, N₂O and CO₂ from three adjacent crab aquaculture ponds
72 converted from paddy fields 12 years ago and neighboring paddy fields (PF) in the Tai Lake basin
73 (31°02'N, 120°25'E; Supplementary Figs. 1 and 2) during 2013–2014. Wheat-rice rotation is the
74 typical cropping system in this region. Urea was applied in PF at 150 and 280 kg N ha⁻¹ during
75 wheat and rice seasons, respectively. Crab ponds differed in size and water depth (Supplementary
76 Table 1). They were not equipped with aerators but were fertilized during culturing. Chinese mitten
77 crab (*Eriocheir sinensis*) were fed with commercial feed pellets, trash fish and corn seeds at the
78 same rate in each pond during crab production period from March to October. Annual C and N
79 inputs in crab ponds were 1.20 Mg C ha⁻¹ and 244 kg N ha⁻¹, respectively (Supplementary Tables
80 2–4).

81 Annual CH₄ emission in PF was 218 ± 7.28 kg CH₄ ha⁻¹ (Fig. 1a), which was located in the
82 upper end of the previously reported ranges (98.3–240 kg CH₄ ha⁻¹) for paddy fields without
83 organic amendment in this area²⁰. However, conversion from PF to crab ponds sharply increased

84 CH₄ emission to 962 ± 62 kg CH₄ ha⁻¹; this value was higher than the summarized amount of 572
85 kg CH₄ ha⁻¹ in permanently inundated temperate wetlands and the default emission factor (900 kg
86 CH₄ ha⁻¹) for tropical inland freshwater wetlands proposed by the Intergovernmental Panel on
87 Climate Change (IPCC; ref. 21).

88 The CH₄ EF_C of C inputs from feeds and fertilizer in crab ponds was estimated at up to 60.0%
89 (Table 1), which may be attributed to the enhanced availability of labile organic substrates and
90 highly anaerobic environment in crab ponds. The C output as harvested crab was 0.19 Mg C ha⁻¹
91 (Supplementary Table 4), accounting for 16.1% of the C inputs excluding the photosynthates by
92 submerged macrophytes. The remaining 1.04 Mg C ha⁻¹ was deposited into sediments as
93 unconsumed feed and feces, which led mean dissolved organic C (DOC) concentrations in pond
94 sediments to reach 7.97-fold greater than that of PF (Supplementary Table 1). Additionally, organic
95 compounds in feed remnants and feces such as starch and protein can be more easily decomposed²²
96 to methanogenic substrates than crop residues in PF. Moreover, pond sediments were permanently
97 inundated, thereby creating anaerobic environments ideal for methanogenesis.

98 Annual N₂O emission in PF was 7.11 ± 0.23 kg N₂O ha⁻¹ (Fig. 1b). The EF_N of fertilizer-N
99 applied was 1.05%, closing to the IPCC default value (1.00%) for agricultural soils²³. Conversion
100 from PF to crab ponds significantly decreased annual N₂O emission by 95.4% to 0.33 ± 0.07 kg
101 N₂O ha⁻¹, with an EF_N of 0.09 ± 0.02% (Table 1). The lower N₂O emission in crab ponds was
102 disproportionate to the differences in N application rates (244 vs 430 kg N ha⁻¹), let alone the
103 relatively higher total inorganic N content in pond sediments (Supplementary Tables 1). Nitrous
104 oxide is derived from both nitrification and denitrification, although denitrification produces more
105 N₂O (ref. 24). It is likely that the much lower redox potential (-124 to -160 mV) suppressed
106 nitrification in pond sediments, which reduced overall NO₃⁻ concentrations to <1 mg N kg⁻¹. This

107 concentration was lower than the threshold value of 5 mg N kg^{-1} for active denitrification²⁵.
108 Moreover, the high DOC concentrations and anaerobic conditions permit N_2O to be further reduced
109 to N_2 through denitrification²⁶.

110 Using the net ecosystem C balance method, annual loss of soil organic C (SOC) in PF was
111 estimated to be $0.04 \pm 0.05 \text{ Mg C ha}^{-1}$ (Table 1), which fell in the range of -0.27 to $0.67 \text{ Mg C ha}^{-1}$
112 estimated previously in paddy fields of Tai Lake basin²⁷. The CO_2 fluxes measured by transparent
113 chambers in crab ponds were regarded as net ecosystem exchange. On an annual basis, crab ponds
114 were weak net CO_2 sources, releasing $0.13\text{--}1.99 \text{ Mg CO}_2 \text{ ha}^{-1}$ (Fig. 1c).

115 Conversion from PF to extensive crab ponds increased the 100-yr GWP from 8.15 ± 0.43 to
116 $28.0 \pm 4.1 \text{ Mg CO}_2 \text{ eq ha}^{-1}$, mainly due to increased CH_4 emission with a contribution of 96.3%
117 (Table 1). Our results contrast with those of Liu et al.¹⁸, who reported such conversion significantly
118 reduced CH_4 and N_2O emissions by 48% and 56%, respectively. Annual CH_4 emission in Liu's
119 ponds (equipped with aerators and classified as semi-intensive, see below) was just $32.6 \text{ kg CH}_4 \text{ ha}^{-1}$
120 despite the much greater feeding rate and higher sediment DOC concentration compared to test
121 extensive ponds. Hence, substrate availability was not the limiting factor for CH_4 emissions in
122 feeding aquaculture systems, while oxygen exposure by aeration was the key factor affecting CH_4
123 emissions. Our results highlight that GHG emissions clearly differ from one aquaculture system to
124 another, greatly depending on the intensity of operational management. This observation illustrates
125 the potential for mitigating the effects of future paddy field conversion through careful
126 management.

127

128 **Global CH_4 and N_2O budgets of freshwater aquaculture**

129 Here, we classified aquaculture into four systems: rice-fish, extensive, semi-intensive and intensive

130 based on local conditions and aquaculture facilities especially whether aerators are used or not (see
131 Methods). We compiled a worldwide database of CH₄ and/or N₂O emissions that were measured in
132 45 inland freshwater aquaculture systems during 2003–2015 (Supplementary Table 5). Land-use
133 and production statistics were also compiled for different aquaculture systems of top 21 freshwater
134 aquaculture producers (Supplementary Table 6); however, data from extensive and semi-intensive
135 systems were pooled because of unavailability of aerator-use data for separate classification. In
136 2014, the top 21 producers contributed 97.5% of global freshwater aquaculture volume¹.

137 The synthesized data show that CH₄ fluxes ranged from –0.03 to 37.0 mg CH₄ m⁻² h⁻¹ in
138 rice-fish, extensive, and semi-intensive systems. Mean CH₄ flux in rice-fish system was the highest
139 at 12.6 ± 3.9 mg CH₄ m⁻² h⁻¹, followed by extensive and semi-intensive systems (Fig. 2a). The
140 absence of CH₄ flux in intensive systems can be attributed to a combination of continuous aeration,
141 water exchange and a lack of habitats for methanogens⁴. The rice-fish system also had the highest
142 mean N₂O flux followed by semi-intensive and extensive systems (28.4 ± 9.8 and 7.56 ± 3.02 µg
143 N₂O m⁻² h⁻¹, respectively; Fig. 2b). The EF_N and yield-scale N₂O EF (EF_Y) in intensive system
144 were 1.16 ± 0.18% and 2.48 ± 0.42 g N₂O kg⁻¹ yield, respectively, which were significantly higher
145 than the corresponding values in extensive (0.24 ± 0.10% and 0.66 ± 0.22 g N₂O kg⁻¹ yield) and
146 semi-intensive (0.35 ± 0.16% and 0.88 ± 0.41 g N₂O kg⁻¹ yield) systems. The EF_Y in intensive
147 systems was close to the IPCC default EF_Y (2.66 g N₂O kg⁻¹ yield) that is widely used in model
148 estimate for aquaculture^{10,21}, but was 2.75- and 1.82-fold greater than that for extensive and
149 semi-intensive systems, respectively. Considering the large volume of extensive and semi-intensive
150 aquaculture (Supplementary Table 6), previous estimates^{10,14} of global aquaculture N₂O emission
151 may have been overestimated because of the higher default EF_Y mentioned above.

152 The estimated CH₄ and N₂O emissions from the top 21 producers in 2014 were 6.04 ± 1.17 Tg

153 CH₄ and 36.7 ± 6.1 Gg N₂O, respectively (Table 2), which accounted for 1.82% and 0.34% of
154 global anthropogenic CH₄ and N₂O emissions, respectively. Methane was a key contributor (94.6%;
155 Fig. 2e) to GWP in freshwater aquaculture, of which 1.19 ± 0.27 Tg CH₄ was emitted from rice-fish
156 system and 4.85 ± 1.04 Tg CH₄ from extensive plus semi-intensive systems. To our knowledge, this
157 is the first global estimate of CH₄ emission from freshwater aquaculture. Our estimated total N₂O
158 emission was much lower than the previous estimates of 90 Gg N₂O (ref. 14) and 146 Gg N₂O (ref.
159 10) of global aquaculture. Extensive plus semi-intensive systems contributed 87.0% of global
160 volume of freshwater aquaculture, meanwhile, were the largest CH₄ and N₂O emitter (80.3% and
161 45.2%, respectively) from this sector. Intensive systems accounted for 8.89% of the production,
162 27.0% of total N₂O emissions but negligible CH₄ emissions. Rice-fish systems represented only
163 4.30% of aquaculture volume, yet they accounted for 19.7% and 27.8% of CH₄ and N₂O budgets,
164 respectively.

165 The greenhouse gas intensity (GHGI, GWP/yield) was 3.59 ± 0.74 kg CO₂ eq kg⁻¹ yield in
166 extensive plus semi-intensive systems, which was 4.46-fold greater than that in intensive systems
167 (0.66 ± 0.11 kg CO₂ eq kg⁻¹ yield; Fig. 2f). Therefore, if half of the current productions from
168 extensive plus semi-intensive systems (19.5 Mt) are replaced by intensive systems, the GWP of CH₄
169 and N₂O emissions from freshwater aquaculture (excluding rice-fish) will be reduced by 40.1%
170 from 143 Tg CO₂ eq to 85.6 Tg CO₂ eq.

171 China has emerged as the world's largest freshwater aquaculture emitter of CH₄ (4.10 ± 0.10 Tg
172 yr⁻¹) and N₂O (22.8 ± 7.1 Gg yr⁻¹), contributing 68.0% and 62.1% of global budgets from the sector,
173 respectively. In China, CH₄ emissions from freshwater aquaculture with 7.57 × 10⁶ ha equates to
174 36.5% of total CH₄ emissions from paddy fields, natural wetlands and lakes (11.3 Tg CH₄ yr⁻¹; ref.
175 28). Since 83.0% of Chinese freshwater aquaculture CH₄ emissions originate from extensive plus

176 semi-intensive systems, a substantial reduction in emissions could be achieved through improved
177 management practices, such as installing more efficient aerators in earthen ponds and implementing
178 optimized feeding strategies for reducing feed waste.

179 In conclusion, the conversion of paddy fields to extensive crab aquaculture ponds sharply
180 increased GWP, primarily through a drastic increase in CH₄ release. Our findings emphasize the
181 need to assess the climatic impacts of land-use shifts towards industrial-scale aquaculture. Methane
182 is the most important GHG in freshwater aquaculture compared with N₂O, and it was primarily
183 sourced from extensive plus semi-intensive systems. Our findings indicate that effective
184 management of extensive and semi-intensive systems through conversion to intensive systems is
185 urgently required to mitigate GHG emissions from the unprecedented growth of aquaculture.

186

187 **Methods**

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

190

191 **Data availability**

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

194

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252

253 **Additional information**

254 **Correspondence and requests for materials** should be addressed to W.D. or C.F.

255

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261

262 **Author contributions**

263 W.D., J.Y., and D.L. designed the study; J.X. led the GHG fluxes and auxiliary measurements with
264 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.
265 and C.F. were the key international collaborators during this research. The manuscript was drafted
266 by J.Y., H.K., W.D. and C.F with all authors contributing to the final version.

267

268 **Competing financial interests**

269 The authors declare no competing financial interests.

270 **Methods**

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

278 Soil was developed from alluvial sediments of the Yangtze River, and classified as stagnic
279 Anthrosols based on the USDA soil taxonomy. The surface soil (0–20 cm) had a pH (H₂O) of 5.95,
280 bulk density of 1.25 g cm⁻³ and a loam texture with 40% sand, 34% silt and 26% clay, and
281 contained 20.3 g kg⁻¹ organic C and 1.81 g N kg⁻¹ total N. Three neighboring aquaculture ponds for
282 Chinese mitten crab (*Eriocheir sinensis*) cultivation were converted from paddy fields in 2001.

283

284 **Experimental design and field management** Four independent 3 × 8 m² plots were established in
285 PF in November 2012. Winter wheat (*Triticum aestivum* L., Yangfumai 4) and summer rice (*Oryza*
286 *sativa* L., Wuyunjing 23) was rotated during the period from November 2012 to May 2014. During
287 wheat season, urea was applied at the rate of 150 kg N ha⁻¹, with basal and supplemental
288 fertilization ratio of 40%:60%. During rice season, urea was applied at the rate of 280 kg N ha⁻¹,
289 with the basal and supplemental fertilizer ratio of 50%:50%. Calcium superphosphate (40 and 125
290 kg P₂O₅ ha⁻¹ for wheat and rice, respectively) and potassium sulfate (60 and 125 kg K₂O ha⁻¹ for
291 wheat and rice, respectively) were applied as basal fertilizers (**Supplementary Table 2**). The row
292 distance was 25 cm for rice and wheat, and the hill distance was 15 cm for rice. No irrigation was

293 performed during wheat season, while rice was managed under a typical water regime mode of
294 flooding-midseason drainage-reflooding-moist irrigation (F-D-F-M). Crop grain and straw were
295 harvested and oven dried at 60°C until a constant weight.

296 Parallel field experiments were conducted in three neighboring crab ponds with different size
297 (CP1, 1.71 ha; CP2, 0.71 ha; CP3, 0.09 ha), from March 2013 to March 2014. Monoculture of
298 Chinese mitten crab was employed at the same stocking density of 15000 ind ha⁻¹ for each pond.
299 The submerged western waterweed (*Elodea nuttallii*) naturally vegetated in ponds and provided
300 molting shelters and foods for crabs. Feeds and fertilizers were applied at the same rates in each
301 pond. Snails (*Bellamya quadrata*) were introduced into the ponds twice at the rates of 600 and 400
302 kg ha⁻¹ on April 4 and June 20, respectively, to filter feed residue and provide supplementary foods
303 for crabs (Supplementary Tables 2 and 3). Crabs were fed with commercial feed pellets (2050 kg
304 ha⁻¹) (Purina Co. Ltd., Jiaxing, China), trash fish (1250 kg ha⁻¹) and corn seeds (1150 kg ha⁻¹),
305 twice per day on 9:00 a.m. and 17:00 p.m. until the crabs were harvested. In order to stimulate
306 phytoplankton and waterweed production, cake manure (residue of de-oiled oil seeds) at 40 kg ha⁻¹
307 was applied as basal fertilizer while urea, compound fertilizer and calcium superphosphate were
308 applied at the rate of 130, 100 and 200 kg ha⁻¹, respectively, with four splits of 25%:25%:25%:25%
309 on March 29, June 5, August 12 and September 27. Annual inputs of C and N to crab ponds were
310 1.20 Mg C ha⁻¹ and 244 kg N ha⁻¹, respectively. Water was constantly maintained all-year round,
311 while the water depth differed between ponds. Crab harvest started from 1 to 30 October 2013,
312 depending on crab maturity. Crab yield was expressed as fresh weight (Supplementary Table 4).
313 Details management practices in the two systems are shown in Supplementary Table 3.

314

315 **Measurement of GHG fluxes** Wooden boardwalks were installed in each plot to facilitate

316 collecting gas samples and measuring the auxiliary parameters (Supplementary Fig. 2). The static
317 closed chamber technique was used to measure GHG fluxes; in PF, PVC chamber collars (50 cm ×
318 50 cm × 20 cm) with a water-filled channel were inserted into the soil at a depth of 15 cm. In crab
319 ponds, a specially designed system, which included four stainless steel pegs for fixing the system
320 and two adjustable crossbars for elevating or lowering the chamber collars with the fluctuation of
321 water level, were installed along the boardwalks to minimize water wave impact on gas sampling.
322 Three PVC chamber collars were placed on the crossbars in each pond. If necessary, the crossbars
323 together with the chamber collars were adjusted to the best position one day before sampling. The
324 transparent Plexiglass chambers (50 cm × 50 cm × 15 cm) in crab ponds and the stainless steel
325 chambers (50 cm × 50 cm × 50 cm) insulated with white foam in PF were used. See Yuan et al.³² for
326 further detailed information of the devices.

327 The GHG fluxes were measured twice weekly in crab ponds during crab production period
328 from March to October and weekly during period without crab production from November to
329 February (Supplementary Fig. 3). In PF, GHG fluxes were measured twice weekly from April to
330 November and weekly from December to March. Gas sampling was conducted at 08:00–10:00 local
331 time to minimize diurnal variation in the flux pattern. During sampling, the chamber was fitted into
332 the water trough of the chamber collars. Each time, four gas samples of the chamber headspace
333 were drawn using a 50-mL syringes at 0, 10, 20, and 30 min after closure and injected into 22-mL
334 pre-evacuated glass vials. Air temperature inside the chamber was simultaneously measured with a
335 mercury thermometer. Concentrations of CH₄, N₂O and CO₂ were determined by a gas
336 chromatograph (Agilent 7890, Santa Clara, CA, USA) equipped with a flame ionization detector for
337 CO₂ and CH₄ and a ⁶³Ni electron capture detector for N₂O. The gas standards were provided by the
338 National Research Center for Certified Reference Materials, Beijing, China. The precision for GHG

339 concentrations was $\pm 0.5\%$ based on repeated measurements of gas standards. The GHG fluxes were
340 calculated using a linear least squares fit to the four points in the time series of concentration for
341 each plot. Data were omitted if the slope of the linear fitting had $R^2 < 0.90$. Since the opaque
342 chambers were used in PF, the measured CO_2 fluxes were ecosystem respiration (R_e); in contrast,
343 CO_2 fluxes in crab ponds measured by transparent chambers were net ecosystem exchange³². The
344 dataset of GHG fluxes were supplied as **Supplementary Table 7**.

345 Annual or seasonal cumulative CH_4 ($\text{kg CH}_4 \text{ ha}^{-1}$), N_2O ($\text{kg N}_2\text{O ha}^{-1}$) and CO_2 ($\text{kg CO}_2 \text{ ha}^{-1}$)
346 emissions (E) were calculated using the following equation:

$$347 \quad E = \sum_{i=1}^n (f_i + f_{i+1}) / 2 \times (t_{i+1} - t_i) \times 24 \times 10^{-2}$$

348 where f represents the flux of CH_4 ($\text{mg CH}_4 \text{ m}^{-2} \text{ h}^{-1}$) or N_2O ($\text{mg N}_2\text{O m}^{-2} \text{ h}^{-1}$) or CO_2 ($\text{mg CO}_2 \text{ m}^{-2}$
349 h^{-1}); i is the i th measurement; $(t_{i+1} - t_i)$ is the days between two adjacent measurements; and $24 \times$
350 10^{-2} was used for unit conversion.

351

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

361 Kjeldahl procedure, respectively³³.

362 **Estimates of SOC change in paddy field and GWP** The SOC change (δ SOC) in PF was estimated
363 from the net ecosystem C balance (NECB) using a coefficient of 0.213 for paddy soils in this
364 study²⁹, namely, the conversion rate of organic C gain to SOC is 213 g C kg⁻¹. The NECB of the
365 short-plant croplands was calculated according to Ma et al.²⁷:

$$366 \text{ NECB} = \text{GPP} - \text{Re} - \text{Harvest} - \text{CH}_4 + \text{Manure}$$

367 where GPP (gross primary production) is inferred from NPP (net primary production) via the
368 NPP/GPP ratio of 0.58 in this region deduced by Zhang et al.³⁴; Re, CH₄ and manure are the C
369 exchange through ecosystem respiration, CH₄ emission, and manure application, respectively;
370 Harvest is the C of removed straw and grain, which was calculated based on biomass yields, and C
371 and N contents in straw and grain (Supplementary Table 4). The NPP includes net primary
372 productions of grain, straw, root, litter and rhizodeposit, according to Ma et al.²⁷.

373 The GWP (Mg CO₂eq ha⁻¹) in PF is calculated by the following equation³⁵:

$$374 \text{ GWP} = 28 \times \text{CH}_4 + 265 \times \text{N}_2\text{O} - 44/12 \times \delta\text{SOC}$$

375 and for crab ponds:

$$376 \text{ GWP} = 28 \times \text{CH}_4 + 265 \times \text{N}_2\text{O} + 1 \times \text{CO}_2$$

377 where CH₄, N₂O and CO₂ denote annual emissions of CH₄ (Mg CH₄ ha⁻¹), N₂O (Mg N₂O ha⁻¹) and
378 CO₂ (Mg CO₂ ha⁻¹), respectively.

379

380 **Data collection and classification of global freshwater aquaculture** As mentioned above, there
381 are large uncertainties in previous model estimates of global aquaculture N₂O emissions by using
382 EFs of applied N and fish yields: First, the N₂O EFs are highly dependent on management levels in
383 the aquaculture system. For example, yield-scale N₂O EF (EF_Y) of carp was 1.07 g N₂O kg⁻¹ yield

384 in an intensive rearing system¹⁶ but was only 0.28 g N₂O kg⁻¹ yield in a semi-intensive earthen
385 pond¹⁷; Secondly, the EF_Y can be biased by the major yield difference between species. For instance,
386 although the direct N₂O emission rates in two adjacent semi-intensive aquaculture ponds were
387 comparable, the EF_Y measured in crab ponds was 8.11-fold greater than that in carp ponds due to
388 the magnitude difference in yields^{17,18}.

389 Here, we compiled a worldwide database of GHG emissions measured in the inland freshwater
390 aquaculture systems (Supplementary Table 5). We identified potential published studies for
391 inclusion in the database using Web of Science with the keywords ‘greenhouse gases or CH₄ or N₂O’
392 and ‘aquaculture or fish farming or rice fish or aquaponics’. Twenty-four studies fell within the
393 inland freshwater aquaculture and met the following criteria: (i) field measurement of CH₄ and/or
394 N₂O emissions was carried out on a per hectare or per fish yield basis; (ii) type of aquaculture
395 system with or without aerator use was reported; (iii) the N input and yield in intensive systems
396 were listed (see below). The dataset include 45 CH₄ and/or N₂O emission measurements across 19
397 sites between 2003 and 2015.

398 Generally, the aquaculture systems are classified based on production per unit volume or per
399 unit area³⁶; however, when estimating the regional or global GHG emissions, such classification
400 might be unfit due to lack of the available production data counted by volume or area and
401 deficiency of the cross-species classification criteria for big differences in production performance
402 between culture species. Here, we classified four systems: rice-fish, extensive, semi-intensive and
403 intensive based on the local conditions and aquaculture facilities especially aerator use or not.
404 Actually, the stocking density and production are associated with investment on infrastructure
405 especially aeration equipment³⁶, because the dissolved oxygen in fish ponds should be
406 maintained >5.0 mg L⁻¹, theoretically³⁷.

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

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

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

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

421 Global inventory of the land use and production statistics are also compiled in different
422 aquaculture systems of the major freshwater aquaculture producers ([Supplementary Table 6](#)),
423 however, data of extensive and semi-intensive systems were pooled because of lack of aerator use
424 data to classify each other. Data were derived from the official fisheries statistics for 2014. In case
425 2014 data were not available, the most recent data were used. If the national official statistical data
426 were not available, the FAO estimate (National Aquaculture Sector Overview) or private survey
427 data were used. Further details on the statistics used are provided in the Supplementary materials.

428

429 **Estimation of global CH₄ and N₂O budgets** We estimated N₂O emissions from intensive systems

430 by multiplying EF_Y by production. Methane emission from intensive systems was recognized as
431 negligible because the aerobic condition limited CH_4 production in such systems⁴. While CH_4 and
432 N_2O emissions from rice-fish, extensive, and semi-intensive aquaculture systems were estimated by
433 multiplying mean emission rates by area, because (i) the yield EF for CH_4 was generally
434 unavailable in literature; and (ii) the EF_Y would be biased by the huge yield difference between
435 species in extensive and semi-intensive systems. Additionally, when estimating CH_4 emission from
436 rice-fish systems, the CH_4 fluxes ($32\text{--}37\text{ mg } CH_4\text{ m}^{-2}\text{ h}^{-1}$) measured in Bangladesh³⁰ were excluded
437 from mean emission rates, because of the extremely high emission rates and relatively small area of
438 rice-fish in Bangladesh ($\sim 3.97\%$ of global rice-fish area).

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

450

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466 **Figure legends:**

467 **Figure 1. Annual CH₄, N₂O and CO₂ emissions from the paddy field (PF) and crab ponds (CP)**

468 **during 2013–2014. a, CH₄, b, N₂O, c, CO₂.** Vertical bars represent standard errors of the means (*n*

469 = 4 for PF and *n* = 3 for crab ponds). Three crab ponds had different size and water depth

470 (Supplementary Table 1). ‘A’, ‘B’, and ‘C’ denote significant differences between sites (*P* < 0.05,

471 ANOVA, Tukey’s HSD test) during the entire year; ‘a’, ‘b’, and ‘c’ denote significant differences

472 between crab ponds during the crab production period or during the period without crab production.

473 CO₂ release from PF was calculated from soil organic C change estimates using the net ecosystem

474 C balance method.

475 **Figure 2. Literature-sourced greenhouse gas emission factors of different aquaculture. a, mean**

476 CH₄ emission rate, **b, mean N₂O emission rate, c, N₂O emission factor of applied N (EF_N), d, yield**

477 based N₂O emission (EF_Y). Boundaries of the boxes indicate the first and third quartiles, line within

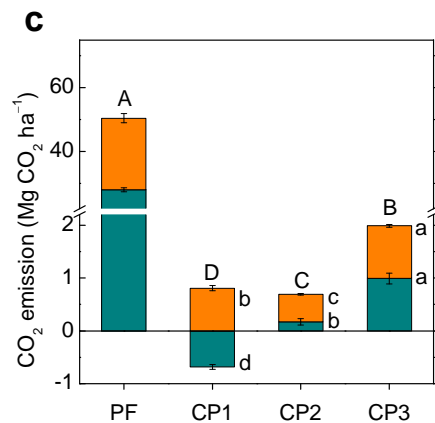
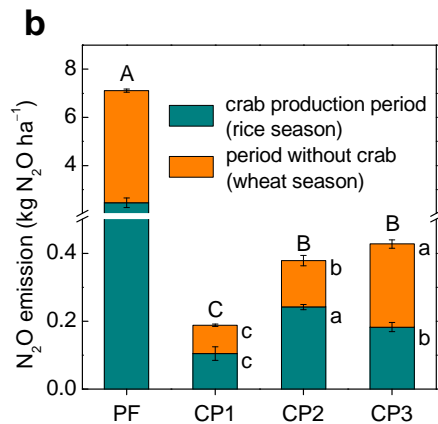
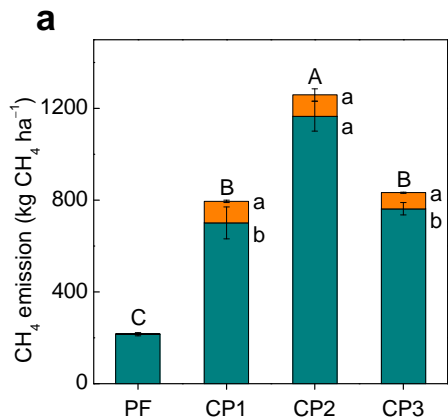
478 the box and the white square represent the median and average, respectively. Whiskers mark the

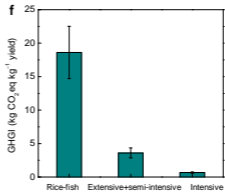
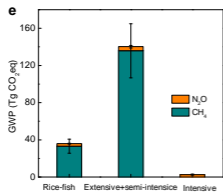
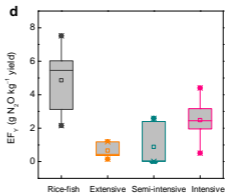
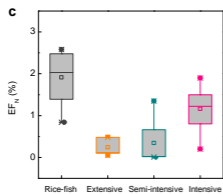
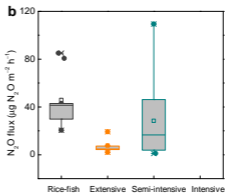
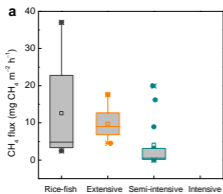
479 10th and 90th percentiles, and the outliers are shown as dots. **e, global warming potential (GWP), f,**

480 greenhouse gas intensity (GHGI, GWP/yield). Vertical bars represent standard errors of the means.

481 Aquaculture systems are classified based on the local conditions and aquaculture facilities

482 especially whether aerators were used or not.





1 **Table 1 Annual GHG emissions, net GWP and emission factors of CH₄ and N₂O in paddy field and crab ponds**

Systems	CH ₄ (kg CH ₄ ha ⁻¹)	N ₂ O (kg N ₂ O 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 ₂ eq ha ⁻¹)	EF _C †† (%)	EF _N ¶ (%)	EF _Y ¶ (g N ₂ O 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

2 * The value is ecosystem respiration in paddy field and net ecosystem CO₂ exchange for crab ponds. † Calculated by application rates and C and
3 N contents of the fertilizers and feeds (see Supplementary Tables 2–4). ‡ Estimated from the net ecosystem carbon balance (NECB) using a
4 coefficient of 0.213 for paddy soils²⁹. § Net GWP = 28×CH₄+265×N₂O–44/12×δSOC for paddy field, and net GWP =
5 28×CH₄+265×N₂O+1×CO₂ for crab ponds. †† The direct emission factor of C for CH₄ (EF_C) is calculated by dividing annual CH₄ emission by
6 total C input²¹. ¶ The direct emission factor of N for N₂O (EF_N) and yield-scaled emission factor for N₂O (EF_Y) are calculated by dividing
7 annual N₂O emission by total N input and grain/crab yield, respectively. Values are means ± standard errors.

8 **Table 2 Direct CH₄ (Gg CH₄ yr⁻¹) and N₂O (Mg N₂O yr⁻¹) emissions from**
 9 **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 ₂ O	CH ₄	N ₂ O	N ₂ O	CH ₄	N ₂ O
	China	696	5,988	3,408	11,653	5,152	3,524
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

10 * Calculated by mean CH₄ and N₂O emission rates (Fig. 2) and the area for aquaculture
11 (Supplementary Table 6) collected from the literature. Rates of CH₄ emission from rice-fish
12 system in Bangladesh were excluded when calculating³⁰. † Calculated by averaged
13 yield-scaled emission factor for N₂O (EF_Y) (Fig. 2d) and volume of production from intensive
14 aquaculture. The direct emission rate of CH₄ from intensive system was estimated at 0
15 according to Hu et al.⁴. ‡ No official or private statistics is available about area and
16 production from different systems in Nigeria.