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# An agronomic assessment of greenhouse gas emissions from major cereal crops

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## Abstract

Agricultural greenhouse gas (GHG) emissions contribute approximately 12% to total global anthropogenic GHG emissions. Cereals (rice, wheat, and maize) are the largest source of human calories, and it is estimated that world cereal production must increase by 1.3% annually to 2025 to meet growing demand. Sustainable intensification of cereal production systems will require maintaining high yields while reducing environmental costs. We conducted a meta-analysis (57 published studies consisting of 62 study sites and 328 observations) to test the hypothesis that the global warming potential (GWP) of $CH_4$ and $N_2O$ emissions from rice, wheat, and maize, when expressed per ton of grain (yield-scaled GWP), is similar, and that the lowest value for each cereal is achieved at near optimal yields. Results show that the GWP of $CH_4$ and $N_2O$ emissions from rice (3757 kg $CO_2$ eq ha$^{-1}$ season$^{-1}$) was higher than wheat (662 kg $CO_2$ eq ha$^{-1}$ season$^{-1}$) and maize (1399 kg $CO_2$ eq ha$^{-1}$ season$^{-1}$). The yield-scaled GWP of rice was about four times higher (657 kg $CO_2$ eq Mg$^{-1}$) than wheat (166 kg $CO_2$ eq Mg$^{-1}$) and maize (185 kg $CO_2$ eq Mg$^{-1}$). Across cereals, the lowest yield-scaled GWP values were achieved at 92% of maximal yield and were about twice as high for rice (279 kg $CO_2$ eq Mg$^{-1}$) than wheat (102 kg $CO_2$ eq Mg$^{-1}$) or maize (140 kg $CO_2$ eq Mg$^{-1}$), suggesting greater mitigation opportunities for rice systems. In rice, wheat and maize, 0.68%, 1.21%, and 1.06% of N applied was emitted as $N_2O$, respectively. In rice systems, there was no correlation between $CH_4$ emissions and N rate. In addition, when evaluating issues related to food security and environmental sustainability, other factors including cultural significance, the provisioning of ecosystem services, and human health and well-being must also be considered.

*Keywords:* $CH_4$, corn, global warming potential, grain yield, maize, meta-analysis, methane, $N_2O$, nitrogen rate, nitrous oxide, rice, wheat, yield-scaled

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## Introduction

Global cropland area is roughly 1.5 billion ha (Thenkabail *et al*., 2010) and the major cereals (rice, wheat, and maize) are produced on approximately 546 million ha (Table 1), representing 36% of this area. These crops provide close to 60% of all human calories, either directly as human food or indirectly as livestock feed (Cassman *et al*., 2003). The Green Revolution and the corresponding intensification of rice (*Oryza sativa*), wheat (*Triticum aestivum*), and maize (*Zea mays*) systems has largely been responsible for averting a short fall in food supply during previous decades (Cassman, 1999; Tilman, 1999; Burney *et al*., 2010). With an expanding world population, the demand for these crops will continue to increase at about 1.3% annually to 2025 (Cassman *et al*., 2003). Broadly, two options exist for increasing cereal production. First, agriculture can be expanded to new areas that are currently not used for food production. Land for such expansion is available especially in Africa and South America (Deininger & Byerlee, 2011). Second, intensification of existing agricultural land can occur by achieving higher yield per unit of land area (Burney *et al*., 2010; Godfray *et al*., 2011), as was achieved in many countries during the Green Revolution. However, because both these options can have negative environmental outcomes, the potential remains for agriculture to further degrade ecosystems in the future. Expansion into new areas can particularly have adverse effects on habitat and biodiversity, whereas agricultural intensification can lead to non-point source pollution and increased greenhouse gas (GHG) emissions (Matson *et al*., 1997; Vitousek *et al*., 1997; Tilman, 1999). Proponents of the latter approach (e.g. Godfray *et al*., 2011) suggest that sustainable intensification, by which higher yields are achieved with no or reduced damage to the environment, will be necessary to meet dual goals of protecting natural resources while ensuring global food security.

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**Table 1** Area in production and average yields of the world's major cereal crops for 2009 (http://www.faostat.fao.org). The "study average yields" are the average yields for each crop in our meta-analysis

| Crop | Total production (Mg) | Harvested area (ha) | Average yields (Mg $ha^{-1}$) | Study average yields (Mg $ha^{-1}$) |
|---|---|---|---|---|
| Rice (paddy) | 678 688 289 | 161 420 743 | 4.2 | 6.05 |
| Wheat | 681 915 838 | 225 437 694 | 3.0 | 4.78 |
| Maize | 817 110 509 | 159 531 007 | 5.1 | 8.01 |

It is estimated that agriculture accounts for 10–12% of total global anthropogenic emissions of GHG, which amounts to 60% and 50% of global $N_2O$ and $CH_4$ emissions, respectively (Smith *et al.*, 2007). $N_2O$ is a more potent GHG than $CH_4$ with a radiative forcing potential that is approximately 12 times larger (IPCC, 2001). Wheat, maize, and other upland crops are primarily a source of $N_2O$ emissions, with these emissions largely driven by the amount of fertilizer N applied (Cole *et al.*, 1997; van Groenigen *et al.*, 2010). Aerobic upland soils contribute little to $CH_4$ emissions and may even be a sink for $CH_4$ in some cases (e.g. Adviento-Borbe *et al.*, 2007). Rice systems are fundamentally different, as rice is typically grown in flooded soils. $CH_4$ is the dominant GHG produced and emitted in these systems, with emissions being largely controlled by water and residue management practices (Yagi *et al.*, 1997; Wassmann *et al.*, 2000). However, rice systems also emit $N_2O$, and it has been shown that the intensity of emissions is related to N fertilizer rate (Zou *et al.*, 2007). In rice systems, there is often an inverse relationship between $CH_4$ and $N_2O$ emissions (Hou *et al.*, 2000). For example, $N_2O$ emissions tend to increase when management practices are implemented to reduce $CH_4$ emissions, through the use of mid-season drains (Cai *et al.*, 1997; Zou *et al.*, 2007). Rice systems are also unique from other systems in that the majority of $CH_4$, as well as some $N_2O$, are emitted through the plant rather than the soil (Yu *et al.*, 1997).

van Groenigen *et al.* (2010) postulated that in a world with increasing food demand and limited land area for expansion of agriculture, $N_2O$ emissions [or global warming potential (GWP)] should be assessed as a function of crop yield (i.e. $N_2O$ produced per unit of grain yield – termed yield-scaled GWP), rather than as a function of area, as is often reported. Ideally, strategies should be identified that allow for the lowest yield-scaled GWP. As GHG emissions are largely driven by fertilizer additions (which tend to increase yield), promoting management practices with low GWP per unit of land area can lead to lower yields. Recently, field study (Ma *et al.*, 2010a; Hoben *et al.*, 2011) and meta-analysis results (van Groenigen *et al.*, 2010) have shown that yield-scaled $N_2O$ emissions were lowest in intensive crop production systems, where crops were grown close to their yield potential with high N use efficiency. These studies reported that a significant increase in yield-scaled emissions only occurred at high or excessive N rates.

Not only is fertilizer N a major driver of $N_2O$ emissions, it is often the limiting nutrient for crop production and therefore is a major driver of crop yields. Although higher yields can often be obtained with greater fertilizer N inputs, the question is whether the yield increase is large enough to offset the corresponding increase in $N_2O$ emissions and result in an overall lower yield-scaled GWP. In rice systems, the relationship between fertilizer rate and GWP is potentially more complex, as $CH_4$ emissions are not as closely linked to N fertilizer inputs as $N_2O$ emissions. Although rice systems have been identified as a substantial source of $CH_4$ emissions, the radiative forcing potential of $CH_4$ is only 8% of $N_2O$. Therefore, we tested the hypotheses that (i) yield-scaled GWP estimates are similar for rice, wheat, and maize and (ii) for each cereal the lowest yield-scaled GWP value is achieved at near optimal yield conditions.

## Materials and methods

### *Data*

An exhaustive literature survey of peer-reviewed publications was carried out using ISI-Web of Science and Google Scholar (Google Inc., Mountain View, CA, USA) for articles published before January 2011. The literature survey focused on GHG emissions from rice, wheat, and maize systems. Studies had to meet specific criteria to be included in the dataset. First, GHG fluxes must have been measured under field conditions for an entire season. A season included the period from planting to harvest. Several exceptions were made when the portion of the season not included in the study was assumed to contribute only a small percentage to overall seasonal emissions (Majumdar *et al.*, 2002; Parkin & Kaspar, 2006; Datta *et al.*, 2009). All wheat studies were conducted on winter wheat except one (Malhi & Lemke, 2007). In temperate climates, winter wheat is planted in the fall and harvested in the summer, covering almost a full year of GHG emissions.

Second, studies on wheat and maize had to report $N_2O$ emissions and studies on rice $N_2O$ and $CH_4$ emissions. Although wheat and maize are potential sources or sinks of $CH_4$, only a few studies (three for wheat and five for maize) measured $CH_4$ emissions (Table 2), all of them determining that the contribution of $CH_4$ to the GWP was minor in these systems (e.g. Halvorson *et al.*, 2010). Although soil $CO_2$ fluxes also represent a source of GHG emissions, on a global scale, they are largely offset by high rates of net primary productivity and atmospheric $CO_2$ fixation by crop plants, and are therefore estimated to contribute <1% to the GWP of agriculture (Smith *et al.*, 2007). Therefore, $CO_2$ as a contributor to GWP was not included in our analysis. Third, only studies that reported crop yields were included. In some cases, grain yield data were obtained from other publications or via personal communication (Table 2).

The GWP of $N_2O$ and $CH_4$ emissions was calculated in units of $CO_2$ equivalents ($CO_2$ eq) over a 100-year time horizon. A radiative forcing potential relative to $CO_2$ of 298 was used for $N_2O$ and 25 for $CH_4$ (IPCC, 2001). We calculated the combined GWP for $N_2O$ and $CH_4$ emissions for each individual rice study prior to meta-analysis. To calculate this value for wheat and maize, we added the average GWP for $N_2O$ emissions across studies to the average GWP of $CH_4$ fluxes.

For each study treatment, the amount of N added as inorganic fertilizer, manure, or green manure was determined. We did not include the amount of N in the previous crop residues as the majority of studies did not report on how it was managed. Also, we did not include N in crop residues applied during the study, as it was not an external input. We divided the studies into four categories based on N fertilization rates (0–<50, 50–<125, 125–200, and >200 kg N $ha^{-1}$ $yr^{-1}$) to examine the relationship between N addition and GWP.

The final data set consisted of 16 (17 sites, 116 observations), 19 (20 sites, 122 observations), and 22 (25 sites, 88 observations) studies for rice, maize, and wheat, respectively (Table 2). All of the rice studies were from Asia. For maize, the majority of studies were from the North America (12), four were from Asia, and three from Europe. For wheat, 12 studies were from Asia, six from Europe, and four from North America. There were no studies from Africa, South and Central America, the Middle East, or Australia.

### *Data analysis*

For every study, the net seasonal GHG flux for each individual treatment combination was included as a separate data point (observation) in our meta-analysis. To avoid bias toward multi-year studies, observations were averaged over years when experiments were repeated over time.

We performed meta-analyses using a non-parametric weighting function and generated confidence intervals (CIs) on flux measurements using bootstrapping. Studies were weighted by replication and sampling frequency. When multiple observations were extracted from the same experimental site within the same study (i.e. when GHG fluxes were measured for multiple treatment combinations), we adjusted the weights by the total number of observations from that site:

$$w_i = n \times f/o, \tag{1}$$

where $w_i$ is the weight for observations from the $i^{th}$ site and $n$ is the number of field replicates (i.e., plots per treatment combination). The variable '$f$' is a measure of sampling frequency, and is equal to the number of flux measurements per month. To prevent studies with high temporal sampling frequency from being assigned extreme weights, all studies that measured GHG fluxes more than once a week were assigned the maximum value of $f$ = 5. Finally, '$o$' is the total number of flux observations from the ith site. By favoring field experiments that were well replicated and frequently sampled, our weighting approach assigned more weight to more precise flux estimates. Furthermore, our approach ensured that all flux measurements could be included in our analyses without any study dominating the data set. Mean GHG fluxes were estimated as:

$$\bar{U} = \Sigma(U_i \times w_i)/\Sigma(w_i) \tag{2}$$

with $U_i$ as the observation of $N_2O$ or $CH_4$ flux from the ith site, and $w_i$ as before. Mean yields and yield-scaled GWP were calculated using the same approach. We used METAWIN 2.1 to generate these mean flux sizes and 95% bootstrapped CIs (4999 iterations) (Rosenberg *et al.*, 2000). Mean fluxes for categories of studies (i.e. the three types of cereal crop, and the categories based on N fertilization rate within each crop type) were considered significantly different if their 95% CIs did not overlap.

To assess the potential for reducing yield-scaled GWP, we identified the treatment with the lowest yield-scaled GWP for each site, and repeated all meta-analyses on the reduced data sets.

We tested whether net seasonal GHG fluxes were correlated with N rate using a simple unweighted regression analysis in SPSS v. 19 (SPSS Inc., Chicago, IL, USA).

We assessed the effect of drainage treatments on yield and GHG fluxes from rice systems by selecting only the subset of experiments that included side-by-side comparisons between continuously flooded and drained fields and repeating all meta-analyses on the reduced data set ($n$ = 7). We used the natural log (ln$R$) of the response ratio as our effect size (Hedges *et al.*, 1999):

$$\ln R = \ln(D/F) \tag{3}$$

where $D$ is the mean value of yield or GHG fluxes in the drained treatment and $F$ is the mean value in the flooded treatment. To ease interpretation, the results for the analyses on ln$R$ were back-transformed and reported as percentage change under drained conditions relative to flooded conditions ([$R$−1]*100). Treatment effects were considered significant if the 95% CI did not overlap with zero.

## Results

Yields for each crop across sites averaged 6.1, 4.8, and 8.0 Mg $ha^{-1}$, for rice, wheat, and maize, respectively. These values were higher than the reported global averages for these crops, but showed the same overall trend

**Table 2** Studies used in the meta-analysis to determine the GWP associated with greenhouse gas emissions for the major cereals

| Crop | Reference | Location | $CH_4$ | $N_2O$ | seasons of data | o* | N rate (kg $ha^{-1}$) | Grain yield (Mg $ha^{-1}$) | GWP range (kg $CO_2$eq $season^{-1}$) | Water mgmt.† | Experimental treatments |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Rice | Abao *et al*., 2000 | Los Baños, Philippines | Y | Y | 2 | 6 | 30–90 | 5.1–6.0 | 75–915 | CF | Residue mgmt./slow release urea |
| | Bhatia *et al*., 2005 | New Delhi, India | Y | Y | 1 | 6 | 0–120 | 3.4–6.4 | 1027–1679 | Dr | N rate/manure and residue mgmt. |
| | Bronson *et al*., 1997 | Los Baños, Philippines | Y | Y | 3 | 12 | 80–200 | 4.7–6.6 | 174–9812 | CF/Dr | N source/residue mgmt./water mgmt. |
| | Cai *et al*., 1997 | Jiangsu, China | Y | Y | 1 | 5 | 0–300 | 5.2–7.3 | 1360–2289 | Dr | N rate/N source |
| | Datta *et al*., 2009 | Cuttack, India | Y | Y | 1 | 4 | 40 | 3.0–4.5 | 1438–2632 | CF | Variety/fish |
| | Ghosh *et al*., 2003 | New Delhi, India | Y | Y | 1 | 7 | 0–120 | 3.8–5.9 | 624–983 | CF | N source/ENF‡ |
| | Kreye *et al*., 2007 | Beijing, China | Y | Y | 2 | 6 | 225 | 3.3–8.5 | 177–1762 | Dr | Water mgmt./plastic ground covers/mulches |
| | Li *et al*., 2009 | Jiangsu, China | Y | Y | 1 | 4 | 300 | 7.9–9.4 | 1691–2922 | Dr | ENF |
| | Ma *et al*., 2007 | Dapu, Jiangsu, China | Y | Y | 3 | 18/12 | 0–270 | 5.6–8.1 | 685–20891 | Dr | N rate/residue mgmt. |
| | Ma *et al*., 2009 | Dapu, Jiangsu, China | Y | Y | 2 | 6/4 | 270 | 6.6–7.5 | 1455–20734 | Dr | Residue mgmt. |
| | Ma *et al*., 2009 | Xingxiang, Jiangsu, China | Y | Y | 1 | 5 | 300 | 5.3–6.7 | 2254–7393 | Dr | Residue mgmt. |
| | Malla *et al*., 2005 | New Delhi, India | Y | Y | 1 | 7 | 120–132 | 5.9–6.9 | 838–1097 | Dr | ENF |
| | Qin *et al*., 2010 | Jiangsu, China | Y | Y | 1 | 6 | 100 | 4.4–6.5 | 1430–3252 | CF/Dr | Water mgmt./organic vs. mineral N |
| | Shang *et al*., 2011 | Hunan, China | Y | Y | 6 | 36/12 | 0–204 | 2.2–8.8 | 3487–22237 | CF/Dr | N,P,K mgmt./green manure |
| | Towprayoon *et al*., 2005 | Samutsakorn, Thailand | Y | Y | 1 | 4 | 119 | 3.9–4.4 | 4035–6201 | CF/Dr | Water mgmt. |
| | Zhang *et al*., 2010 | Jiangsu, China | Y | Y | 1 | 6 | 0–300 | 8.6–10.2 | 2446–4032 | Dr | Biochar/N rate |
| | Zou *et al*., 2005 | Jiangsu, China | Y | Y | 3 | 10 | 150–479 | 6.6–7.9§ | 1476–5711 | CF/Dr | Water mgmt./N rate/residue mgmt. |
| Wheat | Abao *et al*., 2000 | Los Baños, Philippines | Y | Y | 2 | 2 | 60–90 | 1.1–1.4 | 287–302 | RF | Residue mgmt. |
| | Aulakh *et al*., 2001a | Ludhiana, India | N | Y | 1 | 4 | 0–120 | 1.9–5.1¶ | 1077–1264 | IR | Residue mgmt. |
| | Bhatia *et al*., 2005 | New Delhi, India | N | Y | 1 | 6 | 0–120 | 2.5–4.6 | 168–432 | IR | N rate/manure and residue mgmt. |
| | Bhatia *et al*., 2010 | New Delhi, India | N | Y | 1 | 10 | 0–120 | 2.3–5.6 | 147–409 | IR | ENF/tillage |
| | Chen *et al*., 2008 | Jiangsu, China | N | Y | 3 | 18 | 0–300 | 1.4–8.0‖ | 1217–4349 | RF | N rate/tillage/crop rotation/residue mgmt. |
| | Chirinda *et al*., 2010a | Flakkebjerg, Denmark | N | Y | 1 | 4 | 101–170 | 2.8–7.6 | 332–641 | RF | Organic vs. conventional |
| | Chirinda *et al*., 2010a,b | Foulum, Denmark | N | Y | 1 | 5 | 0–165 | 2.8–9.5 | 295–431 | IR | Organic vs. conventional |
| | Grandy *et al*., 2006 | Michigan, USA | N | Y | 2 | 4 | 56–71 | 3.0–4.4 | 214–2478 | RF | Tillage |
| | Kaiser *et al*., 1998 | Braunschweig, Germany | N | Y | 3 | 9 | 0–210 | 6.2–10.8 | 515–1639 | RF | N rate |
| | Kessavalou *et al*., 1998 | Colorado, USA | Y | Y | 1 | 3 | 0 | 2.4–3.1 | 32–35 | RF | Tillage |
| | Liu *et al*., 2011 | Shanxi, China | N | Y | 1 | 2 | 180 | 5.6–6.1 | 702–749 | IR | Residue mgmt. |

**Table 2** (continued)

| Crop | Reference | Location | $CH_4$ | $N_2O$ | seasons of data | o* | N rate (kg $ha^{-1}$) | Grain yield (Mg $ha^{-1}$) | GWP range (kg $CO_2$eq $season^{-1}$) | Water mgmt.† | Experimental treatments |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | Ma *et al.*, 2010b | Jiangsu, China | N | Y | 2 | 8/4 | 138 | 5.0–7.2 | 740–1522 | RF | Residue mgmt. |
| | Majumdar *et al.*, 2002 | New Delhi, India | N | Y | 1 | 6 | 0–120 | 2.0–4.0 | 351–670 | IR | ENF |
| | Malhi & Lemke, 2007 | Saskatchewan, Canada | N | Y | 1 | 2 | 0–120 | 1.9–3.3 | 47–464 | RF | N rate |
| | Malla *et al.*, 2005 | New Delhi, India | N | Y | 1 | 7 | 120–132 | 4.7–5.3 | 220–309 | IR | ENF |
| | Pathak *et al.*, 2002 | New Delhi, India | N | Y | 1 | 8 | 0–120 | 3.4–5.3 | 144–373 | IR | Water mgmt./ENF/manure |
| | Pathak *et al.*, 2006 | New Delhi, India | N | Y | 1 | 6 | 120–180 | 4.2–4.7 | 175–223 | IR | Residue mgmt. |
| | Ruser *et al.*, 2001 | Scheyern, Germany | N | Y | 2 | 2 | 90–180 | 5.3–7.8 | 1262–1703 | RF | N rate |
| | Wagner-Riddle *et al.*, 2007 | Ontario, Canada | N | Y | 1 | 2 | 60–90 | 6.8–7.6** | 318–387 | RF | N rate |
| | Webb *et al.*, 2004 | Top Kingston, UK | N | Y | 2 | 2 | 180–190 | 4.7–7.5 | 328–562 | RF | Cropping system/annual variation |
| | Webb *et al.*, 2004 | Welbeck, UK | N | Y | 2 | 2 | 180–190 | 4.2–6.0 | 468–515 | RF | Cropping system/annual variation |
| | Webb *et al.*, 2004 | Propagation, UK | N | Y | 3 | 3 | 150–190 | 9.0–10.2 | 515–562 | RF | Cropping system/annual variation |
| | Webb *et al.*, 2004 | Shepard's Gate, UK | N | Y | 3 | 3 | 160–230 | 8.4–10.0 | 515–843 | RF | Cropping system/annual variation |
| | Wei *et al.*, 2010 | Shanxi, China | N | Y | 2 | 10/5 | 0–120 | 1.3–5.4 | 546–2407 | RF | Fertilizer mgmt./manure |
| | Weiske *et al.*, 2001 | Giessen, Germany | Y | Y | 1 | 3 | 180 | 6.4–6.7 | 77–165 | RF | ENF |
| Maize | Adviento-Borbe *et al.*, 2007 | Nebraska, USA | Y | Y | 2 | 10 | 140–310 | 12.0–17.5 | 522–2245 | IR | N rate/crop rotation |
| | Adviento-Borbe *et al.*, 2010 | Pennsylvania, USA | N | Y | 2 | 8/4 | 90–225 | 10.6–15.1 | 468–4167 | RF | N rate/crop rotation/manure |
| | Almaraz *et al.*, 2009 | Quebec, Canada | N | Y | 1 | 4 | 0–180 | 2.8–7.9 | 1072–2580 | RF | N rate/tillage |
| | Guo *et al.*, 2009 | Hubei, China | N | Y | 1 | 6 | 0 | 1.8–3.6 | 136–253 | RF | Alley cropping/cropping system |
| | Halvorson *et al.*, 2010 | Colorado, USA | Y | Y | 2 | 14/7 | 0–246 | 7.9–14.3 | 71–406 | IR | ENF |
| | Khalil *et al.*, 2002 | University Putra, Malaysia | N | Y | 1 | 3 | 150–243 | 1.0–1.4 | 351–1241 | RF | Residue mgmt./manure |
| | Liu *et al.*, 2011 | Shanxi, China | N | Y | 1 | 2 | 210 | 7.5–7.6 | 890–1405 | IR | Residue mgmt. |
| | McSwiney & Robertson, 2005 | Michigan, USA | N | Y | 3 | 27/9 | 0–291 | 3.2–9.4 | 453–4721 | IR/RF | N rate |
| | Mosier *et al.*, 2006 | Colorado, USA | Y | Y | 3 | 22/11 | 0–224 | 4.2–12.7 | 93–1869 | IR | N rate/tillage/crop rotation |
| | Parkin & Hatfield, 2010 | Iowa, USA (1) | N | Y | 1 | 2 | 125 | 10.7–11.8 | 2781–3291 | RF | ENF |
| | Parkin & Hatfield, 2010 | Iowa, USA (2) | N | Y | 1 | 2 | 168 | 12.4–12.7 | 2463–2720 | RF | ENF |

**Table 2** (continued)

| Crop | Reference | Location | $CH_4$ | $N_2O$ | seasons of data | o* | N rate (kg $ha^{-1}$) | Grain yield (Mg $ha^{-1}$) | GWP range (kg $CO_2$eq $season^{-1}$) | Water mgmt.† | Experimental treatments |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | Parkin & Kaspar, 2006 | Iowa, USA | N | Y | 2 | 3 | 215 | 9.7–10.0 | 4490–5389 | RF | Tillage/residue mgmt. |
| | Phillips *et al*., 2009 | North Dakota, USA | Y | Y | 1 | 2 | 70 | 2.5–2.7 | 158–203 | RF | N timing |
| | Qian *et al*., 1997 | Nebraska, USA | N | Y | 2 | 2 | 153–234 | 10.3–10.4 | 712–1072 | IR | High nitrate water |
| | Ruser *et al*., 2001 | Scheyern, Germany | N | Y | 2 | 2 | 65–130 | 8.0–9.5 | 585–998 | RF | N rate |
| | Sehy *et al*., 2003 | Scheyern, Germany | N | Y | 1 | 4 | 125–175 | 8.9–10.5 | 1170–3558 | RF | Site specific fertilization |
| | Venterea *et al*., 2010 | Minnesota, USA | N | Y | 3 | 12/4 | 146 | 6.6–10.5 | 281–1592 | RF | N source/crop rotation |
| | Wagner-Riddle *et al*., 2007 | Ontario, Canada | N | Y | 2 | 4 | 50–150 | 3.9–9.0** | 468–824 | RF | N source and mgmt. |
| | Weiske *et al*., 2001 | Giessen, Germany | Y | Y | 1 | 3 | 160 | 7.7–8.0 | 85–159 | RF | ENF |
| | Yan *et al*., 2001 | Tsukuba, Japan | N | Y | 1 | 4 | 0–250 | 2.0–2.5 | 59–255 | RF | N mgmt./ENF |

*Indicates number of observations. A number following a '/' indicates the number of observations used in analysis after averaging similar treatments across years.
†Water management: Continuous flood (CF); rice with any drainage or aerobic period (Dr); irrigated (IR); rainfed (RF).
‡ENF, enhanced-efficiency N fertilizer (includes nitrification inhibitors, urease inhibitors, slow release fertilizers).
§Yield data: personal communication (R. Sass).
¶Aulakh *et al.* (2001b).
‖Calculated by equation (Chen *et al*., 2008; Fig 2a).
**Jayasundara *et al.* (2007).
Y, yes; N, no.

with maize yields being the highest and wheat the lowest (Table 1). Yields ranged between 2.2 and 10.2 Mg $ha^{-1}$ for rice, 1.1 and 10.8 Mg $ha^{-1}$ for wheat, and 1.0 and 17.5 Mg $ha^{-1}$ for maize (Table 2). The average N rate applied was 172, 115, and 152 kg N $ha^{-1}$ for rice, wheat, and maize, respectively (data not shown). The relatively high average N rate for rice is in part due to 10 of the 17 study sites being from China. Fertilizer N rates in China are known to be excessive (Ju *et al.*, 2009) and in our database, China was the only country that had N rates in excess of 200 kg N $ha^{-1}$ in rice systems (Table 2).

### *GHG fluxes, GWP and yield-scaled GWP for each crop*

The average GWP of $CH_4$ and $N_2O$ emissions was highest for rice systems (3757 kg $CO_2$ eq $ha^{-1}$ $season^{-1}$), which was 5.7 times higher than for wheat (662 kg $CO_2$ eq $ha^{-1}$ $season^{-1}$) and 2.7 times higher than for maize (1399 kg $CO_2$ eq $ha^{-1}$ $season^{-1}$) (Fig. 1). The GWP coefficient of variation across all observations for each crop was high, ranging from 0.98 to 1.13, with values across crops being similar. The range in GWP across all observations for each crop was 75–22 237 kg $CO_2$ eq $ha^{-1}$ $season^{-1}$ for rice, 32–4349 kg $CO_2$ eq $ha^{-1}$ $season^{-1}$ for wheat, and 59–5389 kg $CO_2$ eq $ha^{-1}$ $season^{-1}$ for maize (Table 2).

The GWP of $CH_4$ and $N_2O$ emissions from rice systems was largely determined by $CH_4$ emissions (100 kg $CH_4$–C $ha^{-1}$ $season^{-1}$) that accounted for 89% of the GWP. Three rice studies (Ma *et al.*, 2007, 2009; Shang *et al.*, 2011), all from China, reported extremely high $CH_4$ emissions in excess of 20 000 kg $CO_2$ eq $ha^{-1}$ $season^{-1}$, whereas none of the other studies reported values in excess of 10 000 kg $CO_2$ eq $ha^{-1}$ $season^{-1}$ (Table 2). In the few studies that reported $CH_4$ emissions in wheat and maize, $CH_4$ emissions represented <2% of GWP, and on average these systems were a minor sink for $CH_4$ (−0.3 kg $CH_4$–C $ha^{-1}$ $season^{-1}$). All cropping systems were net emitters of $N_2O$, with rice emitting the least (0.88 kg $N_2O$–N $ha^{-1}$ $season^{-1}$) followed by wheat (1.44 kg $N_2O$–N $ha^{-1}$ $season^{-1}$) and maize (3.01 kg $N_2O$–N $ha^{-1}$ $season^{-1}$).

The yield-scaled GWP was significantly higher for rice (657 kg $CO_2$ eq $Mg^{-1}$) than for wheat and maize (Fig. 2). Despite higher GWP of emissions in maize, the yield-scaled GWP was similar for both wheat (166 kg $CO_2$ eq $Mg^{-1}$) and maize (185 kg $CO_2$ eq $Mg^{-1}$) due to lower wheat yields (Table 1).

### *Methane, nitrous oxide and GWP in relation to N input*

There was no effect of N rate on $CH_4$ emissions in rice (Fig. 3b); however, there was a significant correlation between N input and $N_2O$ emissions for all crops (Fig 3a,c,d), although rice $N_2O$ emissions where roughly 60% of that for wheat and maize. On average, 0.68%, 1.21%, and 1.06% of N applied was emitted as $N_2O$ in rice, wheat, and maize, respectively. Yields for all crops increased with increasing N rate, although to a lesser extent in rice systems (Table 3). The yield-scaled GWP was not affected by N rate in rice systems; however, in maize and wheat systems, it tended to be similar for low and medium N rates, but increased at the highest N rates (although this trend was not always significant).

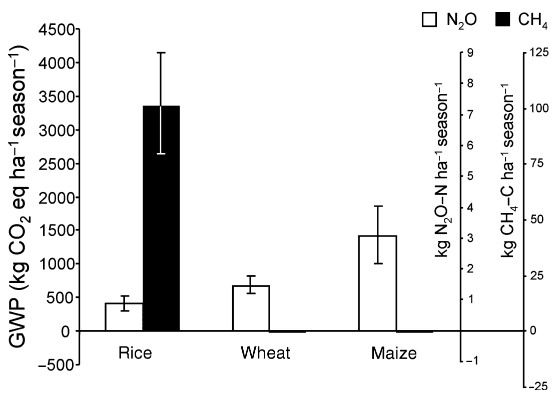


**Fig. 1** Results of a meta-analysis on net seasonal soil fluxes of $CH_4$ and $N_2O$ from three cereal crops. For $N_2O$ fluxes, the results for rice, wheat and maize were based on 116, 122, and 88 observations, respectively. For $CH_4$ fluxes, the results were based on 116, 33, and 8 observations. Observations were weighted by sampling frequency and replication. All error bars represent 95% confidence intervals.

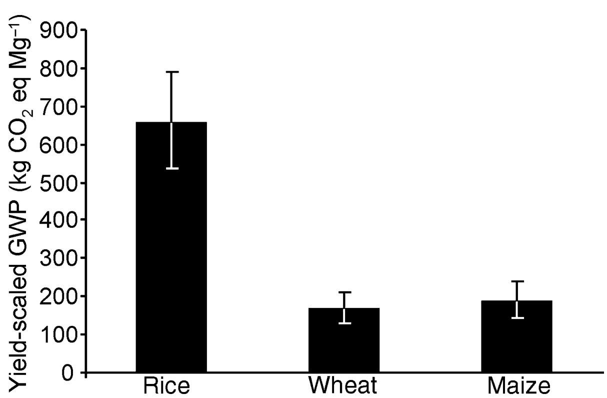


**Fig. 2** Results of a meta-analysis on yield-scaled GWP for three cereal crops. The results for rice, wheat, and maize were based on 116, 122, and 88 observations, respectively. $CH_4$ flux data were available for only a few of the studies on wheat and maize. As $CH_4$ fluxes were negligible for these two crops, they were not included in GWP calculations. Observations were weighted by sampling frequency and replication. All error bars represent 95% confidence intervals.

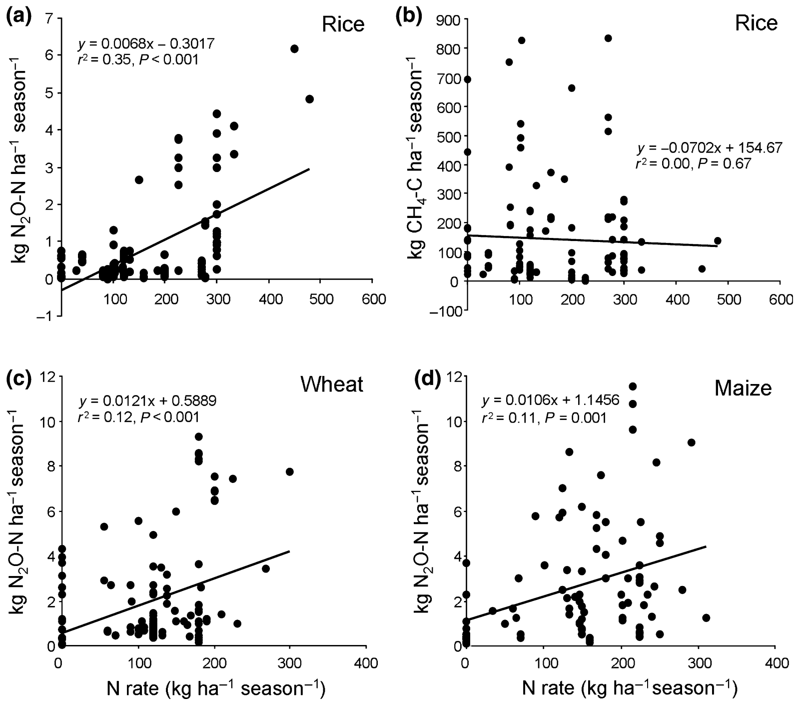


**Fig. 3** Net seasonal soil fluxes of $N_2O$ and $CH_4$ vs. N application rate for rice (a,b), and net seasonal soil fluxes of $N_2O$ vs. N application for wheat (c) and maize (d).

## Discussion

### *Meta-analysis results compared to previous estimates*

To our knowledge, no published estimates of global $N_2O$ emissions exist for individual crops, including the major cereals rice, wheat, and maize. However, a number of studies have attempted to estimate global $CH_4$ emissions from rice systems. Recently, Yan *et al.* (2009) reported that previous estimates have ranged from 25.6 Tg $CH_4$ $yr^{-1}$ (Yan *et al.*, 2009) to 120 Tg $CH_4$ $yr^{-1}$ (Holzapfel-Pschorn & Seiler, 1986), with the IPCC using an estimate of 60 Tg $CH_4$ $yr^{-1}$ (IPCC, 1995). For comparison, we used the mean $CH_4$ emission value from our analysis of 134 kg $CH_4$ $ha^{-1}$ $season^{-1}$ (100 kg $CH_4$–C $ha^{-1}$ $season^{-1}$) and multiplied by the total harvested rice area in (Table 1) to estimate global $CH_4$ emissions of 21.6 Tg $CH_4$ $yr^{-1}$, or roughly one-third of current IPCC estimates. Our value was similar to that of Yan *et al.* (2009), where they estimated that in most Asian rice growing countries, the percent area with one or more drainage events ranged from 57% to 80%. This estimate is roughly in line with our analysis in which 76% of the observations used some form of mid-season drain and/or intermittent irrigation.

For wheat and maize systems, our results indicate that 1.21% and 1.06%, respectively, of applied N is emitted as $N_2O$ (Fig. 3c,d); slightly higher than the suggested 1% used by the IPCC (IPCC, 2006). In flooded rice systems, $N_2O$ emissions resulting from applied N are expected to be lower than for upland crops, and we found that 0.68% of applied N was emitted as $N_2O$. Based on a review, Akiyama *et al.* (2005) reported $N_2O$ fertilizer emission factors of 0.22% for continuously flooded rice systems and 0.37% for intermittently flooded rice systems. The IPCC guidelines estimate that on average 0.3% of N fertilizer applied to rice paddies is emitted as $N_2O$ (IPCC, 2006). The higher values reported from our analysis may be due to the large number of studies from China, where fertilizer N rates for rice systems are generally high (Ju *et al.*, 2009). Such rates were evident in the studies used for this analysis (Table 2). One reason for lower $N_2O$ emissions in rice systems compared with maize and wheat systems is because rice soils are often submerged, so a large portion of the $N_2O$ that is produced is further reduced to $N_2$ (Firestone & Davidson, 1989; Hou *et al.*, 2000; Aulakh *et al.*, 2001a). Most $N_2O$ emissions from rice systems occur during drainage events when $NH_4^+$ is converted to $NO_3^-$, which then becomes susceptible to denitrification (Yao *et al.*, 2010). Indeed, in our analysis, the average $N_2O$ emissions from continuously flooded rice systems was only 0.19 kg $N_2O$–N $ha^{-1}$ $season^{-1}$ (data not shown) compared with the overall rice average that was 0.88 kg $N_2O$–N $ha^{-1}$ $season^{-1}$ (Fig. 1).

**Table 3** Yield, GWP for net seasonal GHG fluxes, and yield-scaled GWP (GWP/yield) for three different cereal crops, as affected by N fertilization rate. As $CH_4$ fluxes were negligible for wheat and maize, GWP for these crops were calculated from net seasonal $N_2O$ fluxes alone

| | N rate | Mean N rate | $o^*$ | Yield | 95% CI | GWP ($CH_4$) | 95% CI | GWP ($N_2O$) | 95% CI | GWP/yield | 95% CI |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | kg N ha$^{-1}$ yr$^{-1}$ | | — | Mg ha$^{-1}$ season$^{-1}$ | | kg $CO_2$eq ha$^{-1}$ season$^{-1}$ | | kg $CO_2$eq ha$^{-1}$ season$^{-1}$ | | kg $CO_2$eq Mg$^{-1}$ | |
| Rice | 0–50 | 13.3 | 17 | 5.14 | 4.29–6.26 | 3124 | 1943–5191 | 151 | 93–209 | 678 | 428–1069 |
| | 50–125 | 109.9 | 46 | 5.24 | 4.96–5.50 | 3117 | 2173–4321 | 173 | 124–240 | 672 | 479–910 |
| | 125–200 | 176.8 | 16 | 6.45 | 6.18–6.74 | 3944 | 2047–6423 | 120 | 75–177 | 630 | 344–982 |
| | >200 | 282.2 | 37 | 7.05 | 6.53–7.53 | 3496 | 2166–5290 | 802 | 585–1049 | 643 | 450–889 |
| Wheat | 0–50 | 0 | 20 | 2.53 | 2.19–2.97 | – | – | 340 | 173–638 | 165 | 78–316 |
| | 50–125 | 107.3 | 58 | 4.48 | 4.07–4.96 | – | – | 573 | 448–743 | 162 | 113–237 |
| | 125–200 | 170.4 | 39 | 6.11 | 5.74–6.50 | – | – | 887 | 650–1191 | 159 | 115–220 |
| | >200 | 252.5 | 5 | 6.27 | 4.44–9.16 | – | – | 1808 | 945–3109 | 369 | 152–602 |
| Maize | 0–50 | 1.6 | 15 | 4.40 | 3.21–6.16 | – | – | 441 | 147–890 | 131 | 39–304 |
| | 50–125 | 90.7 | 11 | 4.13 | 2.91–7.54 | – | – | 527 | 280–1203 | 102 | 85–140 |
| | 125–200 | 150.5 | 31 | 9.31 | 7.40–10.68 | – | – | 1622 | 937–2182 | 183 | 127–232 |
| | >200 | 223.7 | 31 | 9.16 | 7.59–10.80 | – | – | 1827 | 1087–2731 | 246 | 148–374 |

*Indicates number of observations.

*Differences in GWP between crops*

Results from our meta-analysis show that seasonal GWP ha$^{-1}$ is the highest in rice systems, followed by maize and then wheat. Differences in $N_2O$ emissions between maize and wheat are most likely due to differences in N input, which averaged 115 kg ha$^{-1}$ in wheat systems compared with 152 kg ha$^{-1}$ in maize systems. The GWP of GHG emissions for wheat and maize was almost entirely driven by $N_2O$ emissions that are related to fertilizer N input. Although an analysis of the data across all studies support this finding (Fig. 3c,d), data from individual studies show a much stronger correlation between N rate and $N_2O$ emission (e.g. McSwiney & Robertson, 2005; Chen *et al.*, 2008). It is possible that the GWP of $N_2O$ emissions for wheat may be overestimated relative to the other crops in this analysis, as most of the studies were conducted on winter wheat and roughly a third of these winter wheat studies were conducted in temperate climates. As our estimate of GWP covers the period from planting to harvest, winter emissions are included. Some have reported that winter $N_2O$ emissions can account for up to half of annual emissions (Kaiser *et al.*, 1998).

The differences in the GWP of emissions between wheat and maize are small relative to the difference between rice and the other two crops. Rice systems emit $N_2O$, which were related to N input, but $N_2O$ emissions for rice were significantly lower than for either wheat or maize and contributed only 11% to total GWP of rice emissions. The main difference between rice and wheat or maize systems was the high $CH_4$ emissions for rice (Fig. 1). Rice systems are fundamentally different from wheat and maize systems for several reasons. First, rice is typically grown in flooded soils creating anaerobic conditions leading to methanogenesis, which is a strictly anaerobic microbial decomposition process of organic material. Second, rice plants are important conduits of GHGs (both $CH_4$ and $N_2O$) from the soil to the atmosphere (Yu *et al.*, 1997). Methane transport through the plant is the dominant form of methane release into the atmosphere and can account for up to 90% of total emissions (Holzapfel-Pschorn & Seiler, 1986; Butterbach-Bahl *et al.*, 1997). Rice systems are relatively efficient at oxidizing $CH_4$, with an estimated 58% (Sass *et al.*, 1990) to 80% (Holzapfel-Pschorn *et al.*, 1986; Conrad & Rothfuss, 1991) of the $CH_4$ produced in the soil being oxidized by methanotrophs and not emitted to the atmosphere. Interestingly, the high GWP of $CH_4$ and $N_2O$ emissions for rice is despite the fact that 14 of the 17 studies applied some form of drainage (Table 2). As will be discussed later, field drainage is often recommended as a mitigation strategy to reduce $CH_4$ emissions in rice systems.

**Table 4** Results of a meta-analysis on yield, GWP for net seasonal GHG fluxes, and yield-scaled GWP for three different cereal crops. For each experimental site in our dataset, we only considered observations from plots with the lowest yield-scaled GWP. As $CH_4$ fluxes were negligible for wheat and maize, GWP for these crops were calculated from net seasonal $N_2O$ fluxes alone

| Crop | o* | N rate | Grain yield | 95% CI | % of maximum yield† | GWP ($CH_4$) | 95% CI | GWP ($N_2O$) | 95% CI | Yield-scaled GWP | 95% CI |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | kg N $ha^{-1}$ $season^{-1}$ | Mg $ha^{-1}$ $season^{-1}$ | | % | kg $CO_2$ eq $ha^{-1}$ $season^{-1}$ | | | | kg $CO_2$ eq $Mg^{-1}$ | |
| Rice | 17 | 193 | 6.64 | 5.94–7.40 | 94 | 1301 | 797–2011 | 313 | 202–444 | 279 | 169–439 |
| Wheat | 25 | 101 | 5.04 | 4.20–5.87 | 91 | – | – | 595 | 280–1168 | 102 | 61–169 |
| Maize | 20 | 119 | 7.60 | 5.34–9.47 | 90 | – | – | 1115 | 445–1819 | 140 | 82–209 |

*Indicates number of observations.
†Percent of maximum yield was calculated as yield with the lowest yield-scaled GWP/maximum yield obtained per site.

*Yield-scaled GWP: crops and N input*

Focusing on reducing GWP on an area basis, as is commonly done, can be counterproductive, because it can lead to low yielding management practices. In contrast, yield-scaled GWP is an integrated metric that addresses the dual goals of environmental protection and food security. Yield-scaled GWP was significantly higher for rice than either wheat or maize, which were comparable. Therefore, these results lead us to reject our first hypothesis (that yield-scaled GWP for the major cereals are similar). To our knowledge, no reviews have been conducted that compare crops in this manner. However, Pathak *et al.* (2010) recently evaluated the C footprint of Indian food items and reported that yield-scaled GWP of rice was approximately 10 times that of wheat, whereas in our meta-analysis yield-scaled GWP of rice was about four times higher than wheat (Fig. 2). Two of the studies in our analysis (Bhatia *et al.*, 2005; Malla *et al.*, 2005) were from Indian rice–wheat systems, where GHG emissions were measured in both rice and wheat crops. They found that, on average, the yield-scaled GWP of rice was three times higher than wheat (210 kg $CO_2$ eq $Mg^{-1}$ and 73 kg $CO_2$ eq $Mg^{-1}$, respectively, data not shown), suggesting the GWP estimates in Pathak *et al.* (2010) are unexpectedly high and not supported by our findings.

Our data show a trend that yield-scaled GWP for maize and wheat remain relatively constant at low to moderate N rates, but increases when N fertilizer rates are high (Table 3). Other studies have also reported that N rates in excess of crop N demand can lead to large $N_2O$ losses (McSwiney & Robertson, 2005; Ma *et al.*, 2010a) and higher yield-scaled GWP (van Groenigen *et al.*, 2010). In agreement, Hoben *et al.* (2011) recently suggested that maize farmers should be encouraged to apply N at a rate sufficient for maximum economic returns, and that applying N at higher rates to achieve maximum agronomic returns leads to only marginal yield increases, but much higher $N_2O$ emissions.

In rice, the GWP of emissions is primarily driven by $CH_4$ and not $N_2O$. Previous work has shown that increasing the N rate through the use of urea has the potential to both increase (Lindau *et al.*, 1991; Corton *et al.*, 2000) and decrease (Zou *et al.*, 2005; Xie *et al.*, 2010) $CH_4$ emissions. Our study shows no correlation between N rate and $CH_4$ emissions (Fig. 3b) and higher $N_2O$ emissions at excessive N rates (Table 3), leading to a recommendation of achieving maximum economic returns to reduce yield-scaled GWP, similar to wheat and maize.

**Table 5** Summary of the results of the meta-analysis on the effect of drainage on GWP for net seasonal GHG fluxes, yield and yield-scaled GWP (GWP/yield) for rice, using the response metric ln*R* (see Methods). The analysis is based on seven side-by-side comparisons between drained and undrained treatments

| | Average effect (%) | 95% CI |
|---|---|---|
| GWP | −33.6 | −42.9 to −23.7 |
| Yield | 0.95 | −4.69 to 7.63 |
| GWP/yield | −34.3 | −45.6 to −23.4 |
| $N_2O$ | 445 | 121 to 1502 |
| $CH_4$ | −49.5 | −58.8 to −26.3 |

*Opportunities to reduce GWP and yield-scaled GWP*

It is not within the scope of this analysis to identify management practices that result in the lowest yield-scaled GWP for these crops; however, our data set provides insight on the potential to reduce yield-scaled GWP. In rice systems, the average lowest yield-scaled GWP was 279 kg $CO_2$ eq $Mg^{-1}$, whereas in wheat and maize, these values were 102 and 140 kg $CO_2$ eq $Mg^{-1}$, respectively (Table 4). Thus, under this 'best case' scenario, the yield-scaled GWP for rice is about twice as high as for wheat or maize, which is an improvement on the overall average where the yield-scaled GWP was about 3.75 times higher for rice than wheat or maize (Fig. 2).

The yield penalty (i.e. the yield reduction relative to the highest yield for any given site) to achieve the lowest yield-scaled GWP was 6%, 9% and 10% for rice, wheat, and maize, respectively (Table 4). Such results are encouraging when we recognize the need for high yields to achieve food security (Tilman *et al.*, 2002; Lobell *et al.*, 2009; Burney *et al.*, 2010). These data support that of others (Robertson & Vitousek, 2009; Hoben *et al.*, 2011) who have found that substantial $N_2O$ emissions can be avoided by improved N management strategies that better match crop N demand and economic returns, as opposed to applying N for maximum agronomic yields (at N rates which tend to be higher than to achieve maximum economic returns). Such strategies provide a potential win–win for growers and the environment.

In rice systems, draining a field at some point during the growing season is commonly recommended as a means of reducing $CH_4$ emissions (e.g., Yagi *et al.*, 1997; Wassmann *et al.*, 2000). However, draining can also increase $N_2O$ emissions (Hou *et al.*, 2000) which may outweigh benefits to reduction of overall GWP, as $N_2O$ is a much more potent GHG. However, most studies report that the GWP (accounting for both $CH_4$ and $N_2O$) in drained rice fields is lower than in continuously flooded fields (e.g. Zou *et al.*, 2005). Our analysis of studies that included side-by-side comparisons of continuously flooded and drained fields showed that drainage decreased both GWP and yield-scaled GWP by about 34%, but yields were not significantly affected (Table 5). As expected, $N_2O$ emissions were higher and $CH_4$ emissions lower in drained fields. Similarly, Wassmann *et al.* (2000) reported that draining rice fields could mitigate $CH_4$ emissions by 7–80% depending on timing, frequency, and duration of drainage.

*Limitations of our analysis*

Our analysis did not include studies where crops were evaluated side-by-side in the same season (such comparisons are not feasible as these crops have different climatic requirements for optimal production); therefore, direct comparisons among crops were not possible. However, our analysis included a number of studies where one crop followed another in a rotation, meaning the crops were grown on the same soil. An example of this is the rice–wheat rotation (Bhatia *et al.*, 2005; Malla *et al.*, 2005). In these two studies, the GWP of $CH_4$ and $N_2O$ emissions for rice systems was three times higher than wheat, roughly similar to our meta-analytic approach, where it was four times that of wheat.

Most of the studies in our data set were conducted at experimental sites under the management of researchers. Thus, the management and environment may not always be representative of actual production fields. Also, in several of these studies, treatments were evaluated that are not being widely used by farmers. Examples of this include the use of enhanced-efficiency N fertilizer (ENF) (Table 2), plastic mulches (Kreye *et al.*, 2007), and alley cropping (Guo *et al.*, 2009). As most of these treatments were aimed at reducing GWP, it is possible that our overall estimates may be biased toward low GHG fluxes.

Values for annual, rather than seasonal, GHG emissions are needed for these cropping systems as fallow periods can be a large contributor to net GHG fluxes. Moreover, treatment effects like rate of N fertilizer input and its impact on residue yield and C : N ratios can last beyond the growing season. Kaiser *et al.* (1998) reported for a cereal study in Europe that approximately 50% of annual emissions occurred during the winter months (October to February) due to freezing and thawing events during this period. Unfortunately, a meta-analysis of annual GHG emissions was not possible for two reasons. First, the large majority of studies only provided GHG emission data for the growing season. Second, in many parts of the world, especially in the tropics, multiple crops are grown annually on the same piece of land. For example, rice–wheat systems (where rice is grown during the summer and wheat in the winter) are common in both India and China and occupy 24 million ha of land (http://www.rwc.cgiar.org), or about 15% of the total area under rice production. Thus, annual comparisons of GHG emissions from different crops are not always possible, as different crops are grown within the same year.

The important agricultural GHGs are $N_2O$, $CH_4$, and $CO_2$; however, for our analysis, most of the wheat and maize studies did not include measurements of $CH_4$ and $CO_2$. On a global scale, soil $CO_2$ fluxes are largely offset by high rates of net primary productivity and atmospheric $CO_2$ fixation by crop plants, and thus contribute <1% to the GWP of agriculture (Smith *et al.*, 2007). It is generally agreed that the net balance between C respiration and fixation in a cropping system is reflected by changes in soil organic carbon over time (West & Post, 2002; Stewart *et al.*, 2007). However, within a typical study period of 1–2 years (as is the case with most GHG studies), differences in soil organic carbon are difficult to detect as the magnitude of change is small and there is a large degree of spatial variability (Post *et al.*, 2001; Conant *et al.*, 2011). Long-term studies, however, have shown that soil C sequestration or loss can significantly influence net GWP (Robertson *et al.*, 2000; Six *et al.*, 2004) and thus cannot be neglected. In relation to our objectives, reports out of China have found that rice systems have a higher potential for C sequestration than crops grown under aerobic conditions (Pan *et al.*, 2010; Wu, 2011).

To be included in our dataset, wheat and maize studies did not have to report $CH_4$ emissions. Only a few studies measured $CH_4$ emissions and they were minor compared with $N_2O$ emissions. Methane is produced in agricultural soils exclusively by a group of bacteria known as methanogens, via the anaerobic process of methanogenesis that occurs at redox potentials less than −150 mV (Masscheleyn *et al.*, 1993; Mosier *et al.*, 2004). Methanogenesis is a strictly anoxic process and therefore upland cropping systems are not a direct sources of $CH_4$ under normal conditions (Bronson & Mosier, 1993; Robertson & Grace, 2004), although some studies have reported $CH_4$ emissions from upland cropping systems having values as high as 0.6 kg $CH_4$–C $ha^{-1}$ $season^{-1}$ (Abao *et al.*, 2000; Mosier *et al.*, 2006; Halvorson *et al.*, 2010). In contrast, other studies found that the net flux of $CH_4$ in upland soils can be negative, if it is largely a function of $CH_4$ oxidation, where $CH_4$ is oxidized to $CO_2$ by methanotrophic bacteria under oxic conditions. (Robertson *et al.*, 2000; Hütsch, 2001; Robertson & Grace, 2004; Ellert & Janzen, 2008). Although the maximum $CH_4$ oxidation rate in wheat and maize systems was determined to be 2.19 kg $CH_4$–C $ha^{-1}$ $season^{-1}$ for maize (Adviento-Borbe *et al.*, 2007), overall, the effect of $CH_4$ oxidation was minor and represented <2% of GWP.

Finally, the intent of this analysis was not to identify the most sustainable crop with respect to land use decisions and climate change, but rather to compare the GWP of $CH_4$ and $N_2O$ emissions of major cereals based on field GHG measurements and broadly determine if the lowest yield-scaled GWP for each crop was achieved at near optimal yields. Our results support the conclusion that lower yield-scaled GWP values can be achieved at near optimal yields. Addressing the issue of environmental sustainability for each crop is important and new approaches and tools are being developed to more comprehensively assess the impacts of agricultural management decisions using multiple indicators (Steffan-Dewenter *et al.*, 2007; DeFries & Rosenzweig, 2010); however, this was not the objective of this study. Emissions of GHG from soil are only one of many factors that need to be considered when determining the net C footprint of agricultural systems and evaluating potential mitigation practices to be implemented (Smith *et al.*, 2007). Certainly, one other consideration is the C sequestration potential of each system, as discussed above. Another important consideration is the percentage of these cereals that are consumed directly by humans. Rice is the staple crop for the largest number of people on earth, and human consumption accounts for 85% of total production of rice. In contrast, 72% of wheat and only 19% of maize is directly consumed by humans (Maclean *et al.*, 2002). A larger proportion of these crops are used for other purposes such as livestock feed and biofuels, which in turn have associated GHG emissions and energy requirements that would need to be factored into a complete life cycle analysis to more fully understand and account for environmental impacts among cereal crops (Hill *et al.*, 2006; Garnett, 2009). Other considerations that would need to be addressed, but are much more difficult to quantify, are the ecosystem services these crops provide and the roles of these crops in various cultures (Steffan-Dewenter *et al.*, 2007; Sachs *et al.*, 2010). For example, in California, 230 wildlife species and an estimated 10–12 million water birds use rice fields annually (Sterling & Buttner, 2009). It is increasingly recognized that integrated metrics involving ecosystem services, GWP estimates, and other important aspects of agriculture (e.g. human health, food security, economic prosperity, and sociocultural well-being) need to be developed to begin to monitor and compare the impacts of agricultural practices on a global scale (Sachs *et al.*, 2010).

*Need for comparable GHG emission estimates*

When conducting this meta-analysis, it was apparent that most studies only discussed results for GHG emissions relative to other treatments in their study rather than to other comparable studies. One possible reason for this is that, to our knowledge, there are no established or well-accepted values for comparison of GWP results between studies. In this meta-analysis, we made an attempt to provide average values for GWP, calculated per ha and per ton of grain produced (yield-scaled). A second problem is that many different units are used to express GWP that makes straightforward comparisons with other studies difficult. When compiling this data set, a combination of the following units was used: $CH_4$ (or $CH_4$–C), $N_2O$ (or $N_2O$–N) μg, mg, kg, $h^{-1}$ $day^{-1}$, $season^{-1}$, and $year^{-1}$. To allow for better comparisons to be made between values in the future, we propose for field studies that GHG be reported as kg $CH_4$–C (or $N_2O$–N) $ha^{-1}$ $season^{-1}$ (or $year^{-1}$, depending on study) and that yield-scaled GWP be reported as kg $CO_2$ eq $Mg^{-1}$.

Interestingly, the primary limitation to compiling a larger data set for this analysis was the absence of yield data in studies where GHG were measured. As there are concerns about global food security and how mitigation strategies may affect yield, it will become increasingly important to link the intensity of GHG emissions with the primary function of these systems (i.e. producing food). Therefore, when GHG studies are conducted in cropping systems, yield data should be routinely provided to facilitate further and more in-depth analysis on the relationship between mitigation practices, GHG emissions, and yield.

## Conclusions

We conducted a comprehensive meta-analysis of GHG emissions in major cereal cropping systems and related GWP with grain yield. Our results show that the GWP of $CH_4$ and $N_2O$ emissions from rice was significantly higher than for wheat or maize when expressed on an area basis. Likewise, yield-scaled GWP remained about 3.75 times higher for rice compared with wheat and maize. The higher GWP of emissions from rice were largely driven by $CH_4$ emissions, which were unaffected by fertilizer N input. However, of the three cereals evaluated, rice systems showed the greatest potential for reduction in yield-scaled GWP. The lowest yield-scaled GWP values for each crop were achieved at near optimal yields, highlighting the potential for sustainable intensification of these cereals that will be necessary to meet the increased demand brought on by increasing population. There was also evidence that relatively small increases in yield, in particular, for wheat and maize through increased N fertilization, led to disproportionally higher yield-scaled GWP. This finding indicates that better N management practices would allow for a reduction in yield-scaled GWP without significant yield penalties. Our analysis showed that mitigation practices to reduce yield-scaled GWP for rice should be mainly focused on the reduction of $CH_4$ emissions, even though these mitigation practices will probabaly lead to higher $N_2O$ emissions. Importantly, we were not able to evaluate the C sequestration potential of these systems in our analysis and some studies have shown that rice systems have a greater potential to sequester C than upland cropping systems. Finally, GWP is only one of many factors that need to be considered when evaluating the sustainability of different cereal crops. For example, more rice is directly consumed by humans than either wheat or maize, where additional GHG emissions and energy costs may be associated with subsequent conversion of these crops into feed or fuel. In the context of climate change and agriculture, there is growing consensus that other factors including cultural significance, the provisioning of ecosystem services, food security, and human health and well-being must also be considered.

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